DRAFT FOR STAKEHOLDER REVIEW

Review and Synthesis of Available Information to Estimate Human Impacts to Dissolved Oxygen in Hood Canal

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Introduction

Hood Canal is a long, deep fjord-like waterbody in Puget Sound with relatively low human development in the surrounding watershed. The tidal exchange between Hood Canal and Admiralty Inlet is small relative to the overall depth and volume of the canal, and a sill at the north end of Hood Canal restricts circulation. Because of these characteristics, low dissolved oxygen (hypoxia) is a natural condition in the deep waters of Hood Canal.

Fish and other aquatic organisms can be acutely and chronically impacted by the depletion of dissolved oxygen, and periodic fish kills in Hood Canal have been attributed to low dissolved oxygen conditions. Because of the severity of the natural hypoxia, it is important that we understand whether human impacts, even if relatively small, are exacerbating the impacts to fish and other aquatic life. The Washington water quality standards are specifically designed to minimize human-caused impacts to dissolved oxygen in waters with naturally low dissolved oxygen like Hood Canal. The standards require that human impacts be restricted to an impact of less than 0.2 mg/L (interpreted as a non-detectable change to natural conditions) any place in the water column and at any time.

Thus, a key question for both the water quality agencies and the public is "What is the human contribution to the low dissolved oxygen in Hood Canal?" This is a difficult question to answer. An investigation of this question involves analysis of watershed conditions, population and land use development, inputs from septic systems, processes that affect nitrogen transport, and flushing and productivity in the Canal. This review focuses on the introduction of human-caused nitrogen loadings to the surface waters of Hood Canal. These releases stimulate phytoplankton growth, and this in turn alters the oxygen balance in the Canal. While there are many analytical pieces to the puzzle, this assessment is organized around three sequential questions:

1. How much nitrogen do humans and other sources in the watershed contribute to Hood Canal?

2. How much nitrogen do humans contribute to the surface layer of Hood Canal compared to marine sources of nitrogen?

3. What is the impact of human nitrogen contributions on dissolved oxygen in Hood Canal?

The first question provides the foundation for estimating the potential human impacts to dissolved oxygen. The second question provides a gross estimate of the relative significance of human contributions and natural conditions on phytoplankton growth and dissolved oxygen depletion. The third question directly addresses concerns about low dissolved oxygen and fish kill events. It is also the most complicated to answer.

The purpose of this document is to review available scientific information and provide the best estimates of the current human impact to dissolved oxygen levels in Hood Canal. Several regulatory options described in Eaton and Baldi (2012) require an assessment as to

whether human contributions are meeting, nearing, or violating state water quality standards. We consider uncertainty a fundamental part of the assessment, but we also recognize the need to move forward with management actions with the available information. The goals of this document are to (1) accurately describe the extensive research to date in a condensed document, and (2) provide "best estimates" of human impacts to dissolved oxygen based on all available information and our best professional judgment. In some instances, we derive these estimates from methods and data sets that have not been integrated to date.

This review was initiated by a request from the Hood Canal Coordinating Council to support the development of the Aquatic Rehabilitation Action Plan (Brewer, 2011). We reviewed the information available as of April 2011, which included journal articles, published technical reports, and draft chapters available through the Hood Canal Dissolved Oxygen Program (HCDOP) web site (http://hoodcanal.washington.edu/news-docs/publications.jsp). We also received additional information from the lead authors of the referenced studies who reviewed the initial draft report. We held three meetings to discuss findings and areas of agreement and disagreement in June 2011.

Some of the studies reviewed in this report are papers that have been published in peer-reviewed journals or technical reports, while others had not been peer-reviewed prior to the development of this report. This has led to a parallel development of this report and some of the underlying studies. In particular, several chapters of the Hood Canal Dissolved Oxygen Program report were released for the first time during the EPA/Ecology review, and other chapters were revised during the development of this report.

Recognizing the importance, complexity, and interdisciplinary aspects of the scientific questions under review, EPA and Ecology requested that the Puget Sound Institute conduct an additional round of review from an independent panel of experts in early 2012. The panel review was guided by specific charge questions that focused on differences in methodology or interpretation among the researchers. The panel identified a number of important issues in the analyses to date (PSI, 2012). The independent review was shared with all researchers, who provided feedback. This version reflects the independent review as well as ongoing discussions with researchers. We emphasize that peer review generally leads to better scientific products, but it does not always lead to consensus. Some of the findings in this report are subjects of ongoing debate among the researchers.

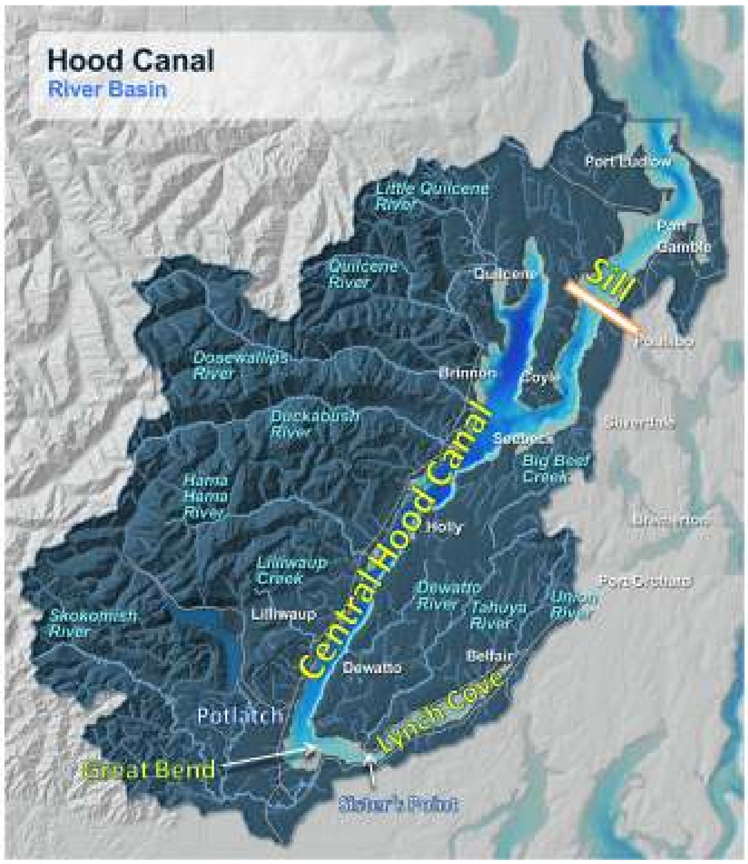
We begin this summary of available science with a conceptual framework of the important system components, including a description of the problem. In turn we address each of the three questions described above, based on the best available published information listed in Table 1 below. Finally, we address uncertainty, identify factors not explicitly addressed in currently available information, and summarize broader findings.

Table 1. Published source documents on Hood Canal reviewed under this synthesis (see Reference section for complete citations).

Study Topic	Analytical Method	Information Reviewed
Question 1: Human Nitrogen Contributions	Water Quality Monitoring	Embrey and Inkpen (1998) Simonds, B., Sheibley, R., Rosenberry, D., Reich, C., and Paulson, A. (2008). Georgeson, A., Mathews, W., and Orth, P. (2008). Banigan, L. (2008) James, A. (2011a)
	Statistical Loading Models	Paulson, A., Konrad, C., Frans, L., Noble, M., Kendall, C., Josberger, E., Huffman, R., and Olsen, T. (2006) Steinberg, P., Brett, M., Bechtold, J., Richey, J., Porensky L., and Osborne, S. (2010). Richey, J., Brett, M., Steinberg, P., Bechtold, J., Porensky L., Osborne, S., Constans, M., Hannafious, D., and Sheibley, R. (2010).
Question 2: Human vs Marine Nitrogen Contributions	Current Meter Data Analysis	Paulson, A., Konrad, C., Frans, L., Noble, M., Kendall, C., Josberger, E., Huffman, R., and Olsen, T. (2006)
	Estuarine Aggregated Models	Steinberg, P., Brett, M., Bechtold, J., Richey, J., Porensky L., and Osborne, S. (2010) Devol, A., Newton, J., Bassin, C., Banas, N., Kawase, M., Ruef, W, Bahng, B., Warner, M. (2011a).
Question 3: Human Impacts on Dissolved Oxygen	Sediment Core Analysis	Brandenberger, J.M., E.A. Crecelius, P. Louchouarn, S.R. Cooper, K. McDougall, E. Leopold, and G. Liu. (2008) Brandenberger, J.M., P. Louchouarn, and E.A. Crecelius (2011)
	Water Quality Monitoring	Bassin, C.J., Mickett, J.B., Newton, J.A., and Warner, M.J. (2011) Warner, M. (2011a) Mickett, J., Alford, M. Newton, J. Devol, A. (2011)
	Aggregated Models	Brett (2010a) Devol, A., Newton, J., Bassin, C., Banas, N., Kawase, M., Ruef, W, Bahng, B., Warner, M. (2011)
	Water Quality Model of Hood Canal	Kawase (2010) Kawase and Bahng (2010) Kawase and Bahng (2012)

Studies compiled for this assessment have used different naming conventions for the sub-regions within Hood Canal. In this review, "Hood Canal" means the entire waterbody and/or watershed south of the sill near Poulsbo on Figure 1. South of the sill, we distinguish between the main axis or main arm of the Canal as "Central Hood Canal" (Poulsbo to Potlatch), a transition area called the "Great Bend" (Potlatch to Sisters Point), and "Lynch Cove" (East of Sisters Point). Many documents refer to the area east of Sisters Point as Lower Hood Canal and consider Lynch Cove only the far eastern extent of that area.

Figure 1. Hood Canal watershed, place names, and naming conventions used in this document.

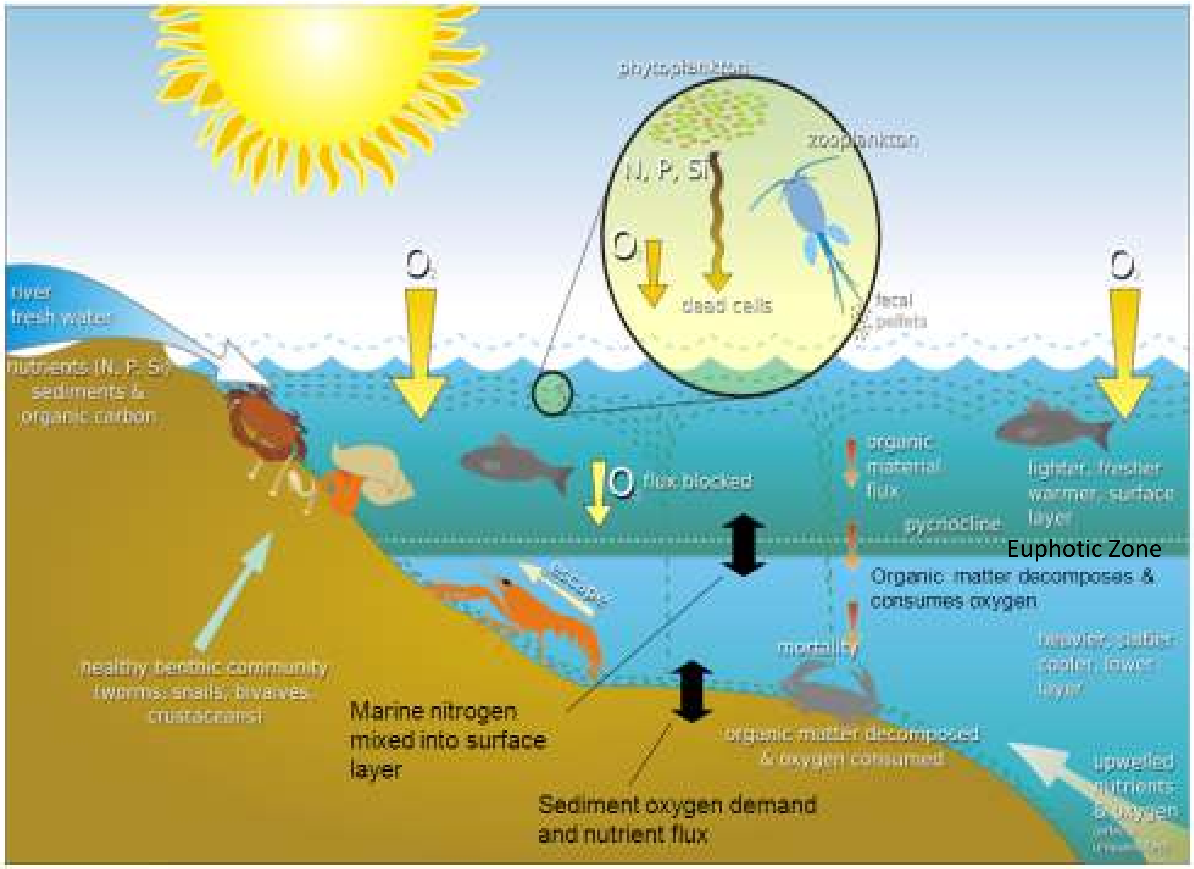


Source: Adapted from graphic on HCDOP Website (www.hoodcanal.washington.edu)

Conceptual Framework

Fortunately, all of the researchers studying Hood Canal hypoxia are operating from similar conceptual models of the system and key attributes related to oxygen. It is not feasible to fit all the potential sources and processes of concern in one easy-to-read graphic, but Figure 2 captures most of the characteristics of concern in Hood Canal. This figure is part of the documentation for studies of hypoxia in the Gulf of Mexico and captures the critical physical, chemical, and biological processes that govern dissolved oxygen in Hood Canal as well.

Figure 2: Conceptual model of processes related to dissolved oxygen.



Source: Adapted from Downing JA, et al. Gulf of Mexico hypoxia: land and sea interactions. Task force report no. 134. Ames, IA:Council for Agricultural Science and Technology, 1999.

Two black double-arrows added to the figure represent important processes in Hood Canal. The first arrow highlights the estuarine mixing of nitrogen-rich water from deeper waters into the surface layer, a natural mixing process that provides nutrients for phytoplankton in the surface layer. The second arrow highlights the processes occurring at the sediment-water interface, where enriched sediments exert an oxygen demand on the overlying water and release dissolved nutrients under low oxygen conditions. A full understanding of human impacts on oxygen must consider the effects of these processes.

Oxygen and Nutrients

Dissolved oxygen conditions in Hood Canal have been monitored extensively by the Washington State Department of Ecology and the University of Washington over the past several decades. The Hood Canal Dissolved Oxygen Program (HCDOP) began an expanded monitoring effort in 2005. Roberts et al. (2005) described the initial monitoring program components. Funded by the U.S. Navy and led by the University of Washington, HCDOP significantly

increased the resources focused on conditions in Hood Canal. HCDOP included several monitoring and modeling programs:

- Marine water quality monitoring (4 ORCA buoys, cruises)
- Flow and water quality in freshwater streams and rivers
- Biotic impacts of low dissolved oxygen
- Phytoplankton productivity sampling and analysis
- Terrestrial models and other analyses
- Marine models and other analyses

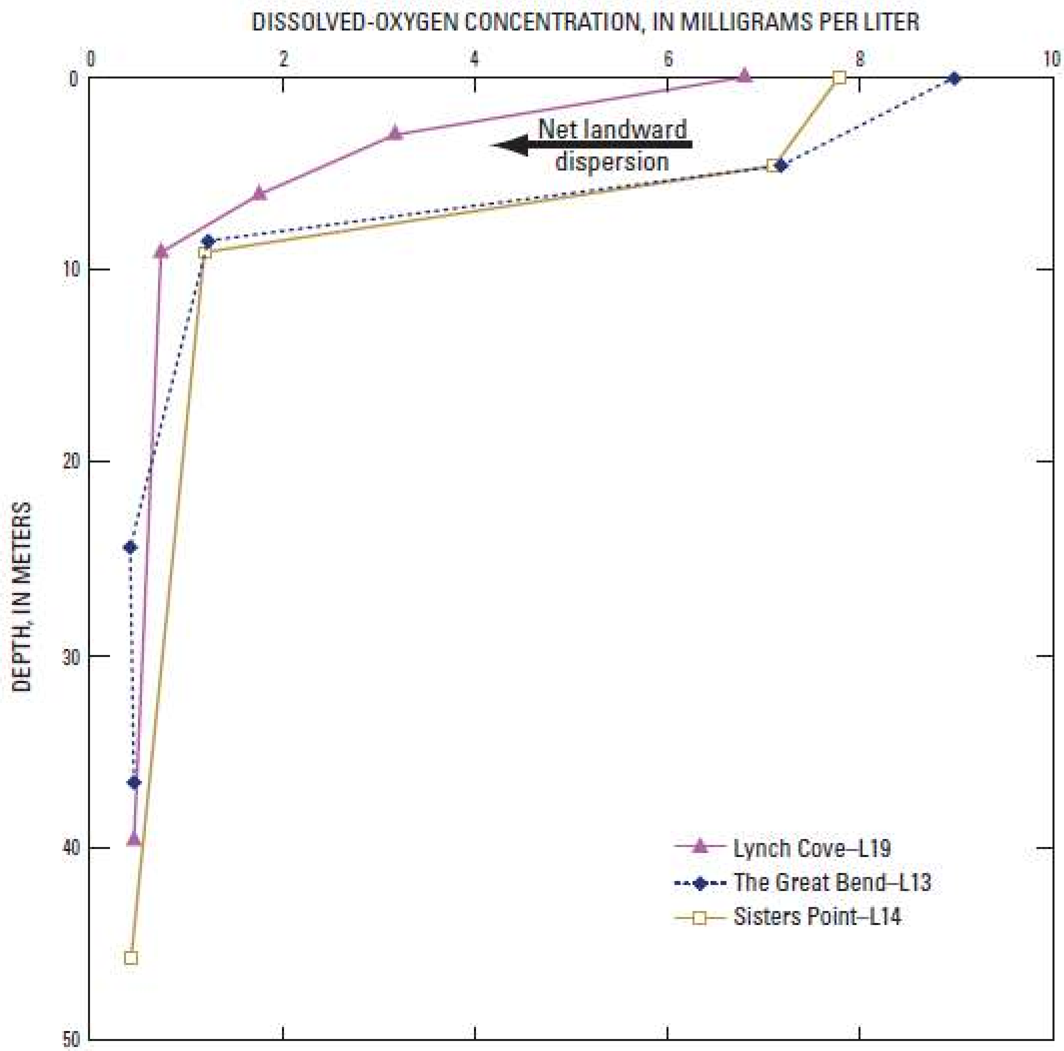
In addition to these activities funded through HCDOP, several other research efforts addressed conditions in Hood Canal, including:

- Sediment cores (Pacific Northwest National Laboratory, Texas A&M University, University of Washington, USGS, and Bryn Athens College)
- Terrestrial, marine, and groundwater nitrogen loads to Hood Canal (USGS)
- Groundwater monitoring within the Hood Canal watershed (Mason County, Kitsap County, USGS)

The resulting body of information for the 2005-2009 intensive-monitoring period provides insight into the spatial and temporal variation in oxygen concentrations along with current nitrogen loadings to Hood Canal.

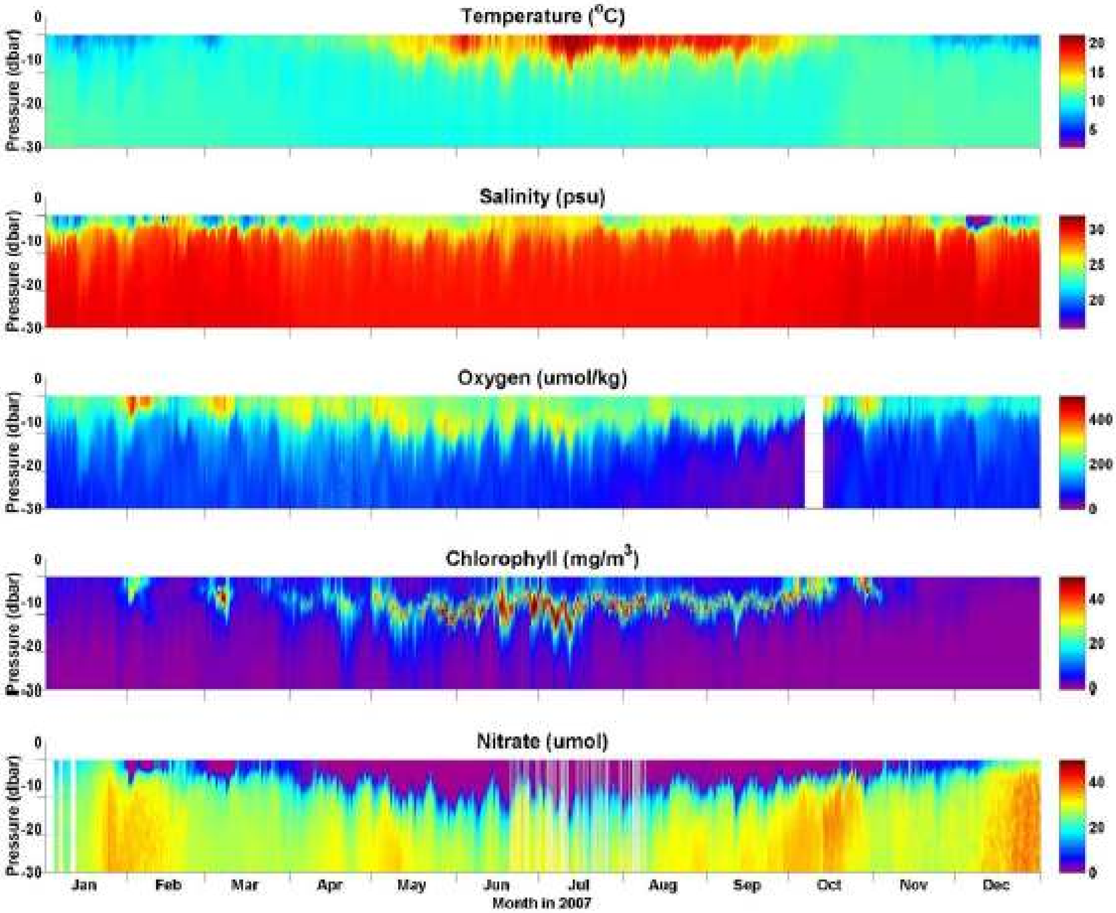
We present several figures to illustrate the types of monitoring information and some of the spatial and temporal patterns of dissolved oxygen in Hood Canal (Figures 3 through 8). Dissolved oxygen levels deeper than 10 meters below the surface in Hood Canal and Lynch Cove are perilously low for fish in the late summer and fall, with a sharp difference between the surface mixed layer and the very low oxygen levels in the lower layer (Figure 3). Concentrations below 1 mg/L can be lethal to some species (EPA, 2000), and Newton et al. (2011c) describes potential chronic and acute effects of low dissolved oxygen on Hood Canal biota. Oxygen levels fluctuate seasonally throughout the water column (Figures 4 and 5). The magnitude of low dissolved oxygen levels varies spatially (Figure 6), and the volume-weighted concentrations exhibit large interannual variability (Figures 7 and 8).

Figure 3. Examples of vertical dissolved oxygen profiles at three locations in Hood Canal in August 2004.



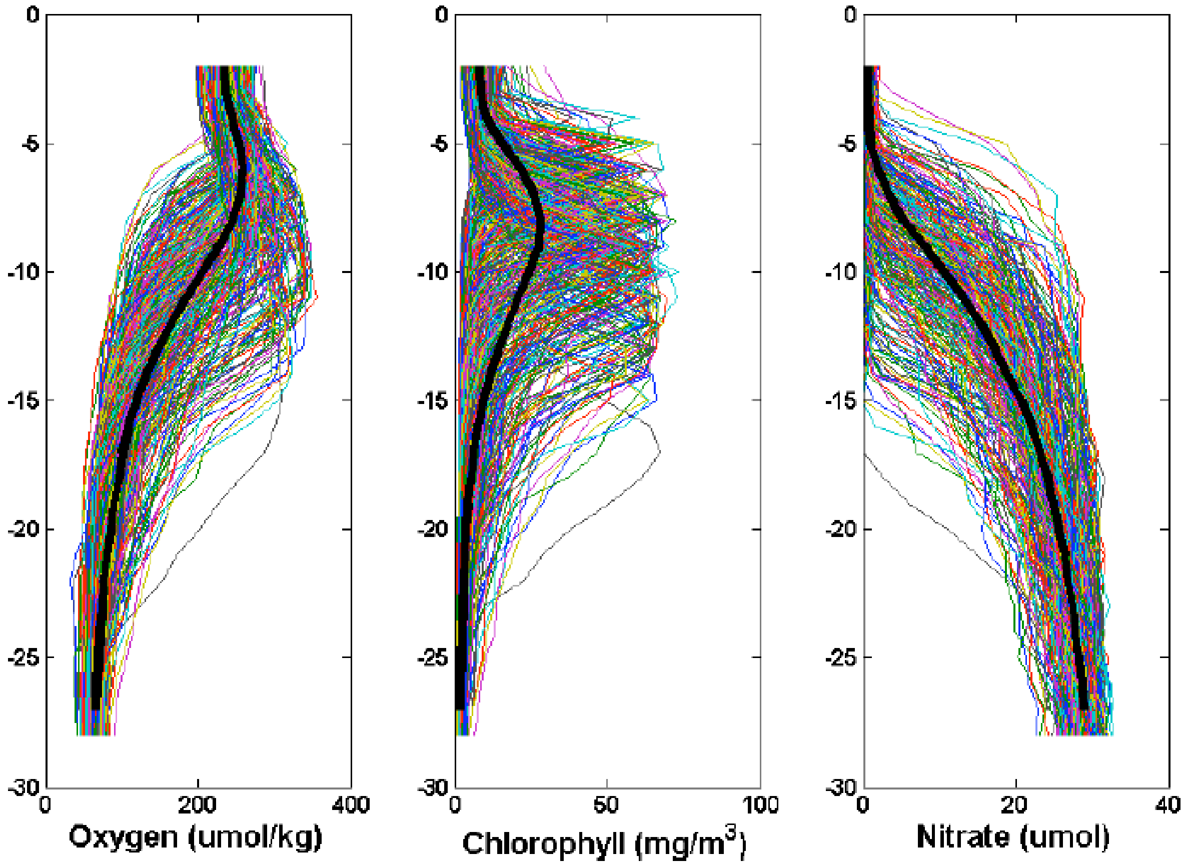
Source: USGS (Paulson et al., 2006)

Figure 4: Continuous water quality data from an ORCA buoy in Lynch Cove



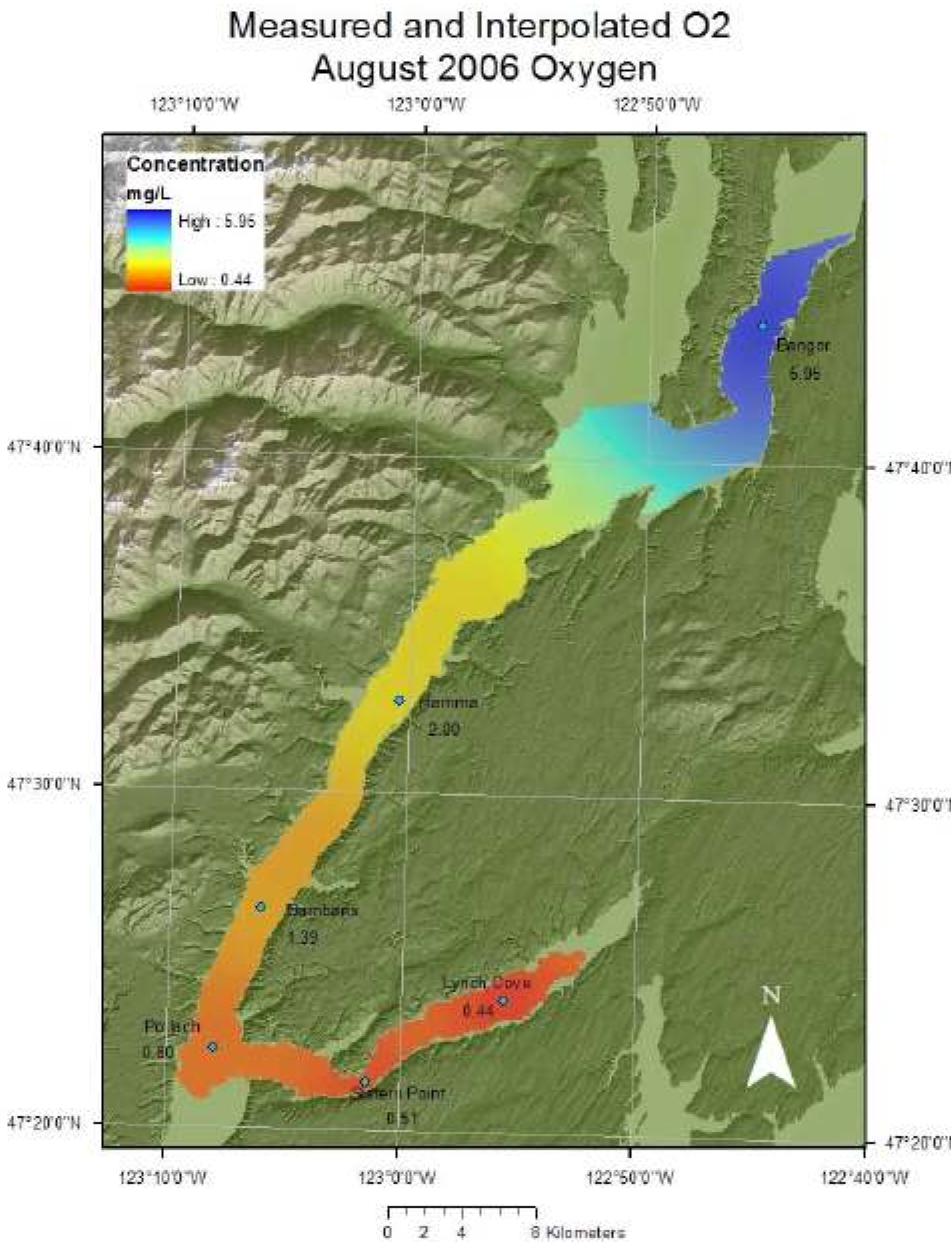
Source: Devol, et al. (2011b)

Figure 5: All measured vertical profiles of oxygen, chlorophyll, and nitrate for July 2008 at the ORCA buoy in Lynch Cove



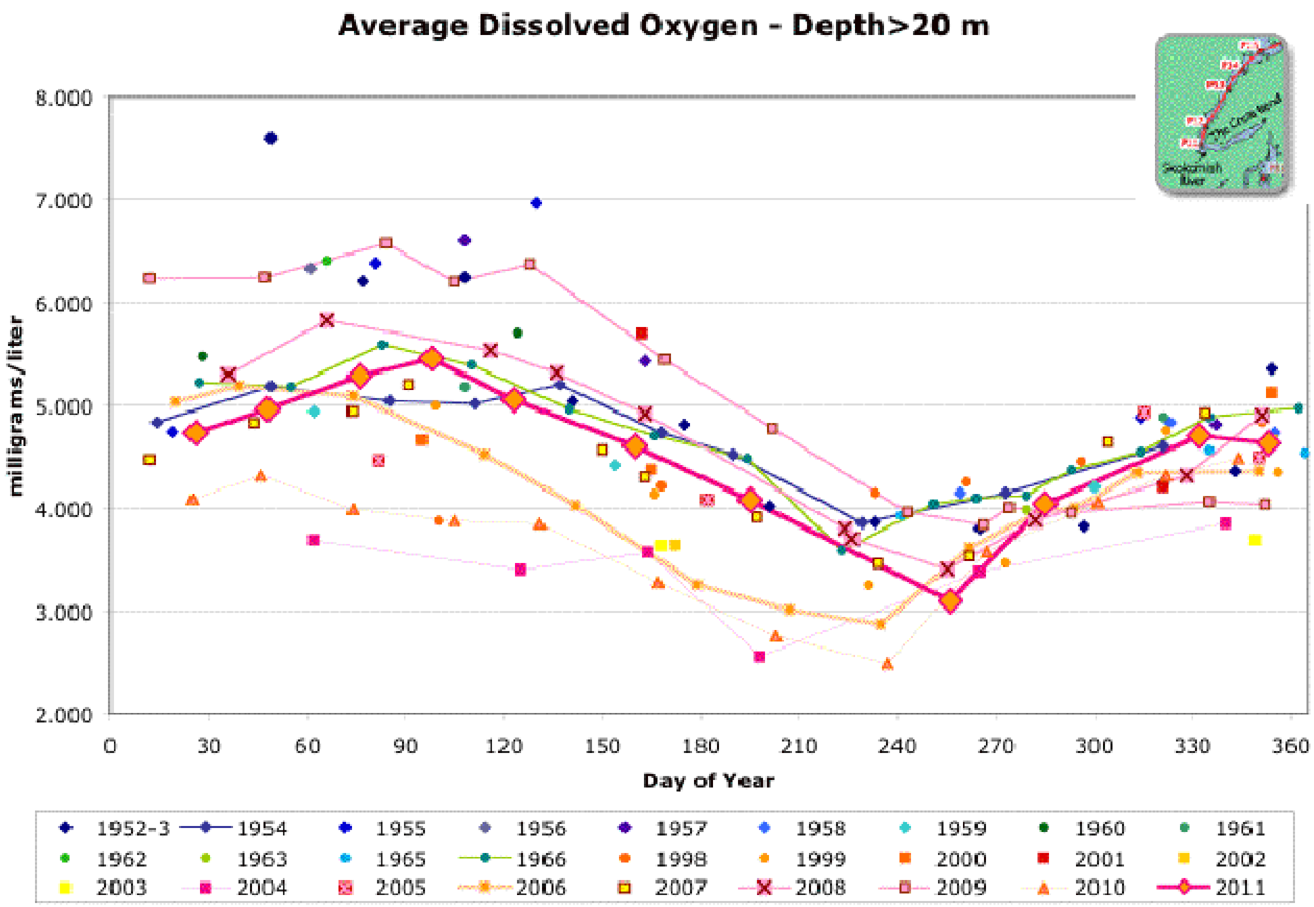
Source: Newton et al. (2011a)

Figure 6: Longitudinal pattern of minimum dissolved oxygen concentrations in Hood Canal (near-bottom depth).



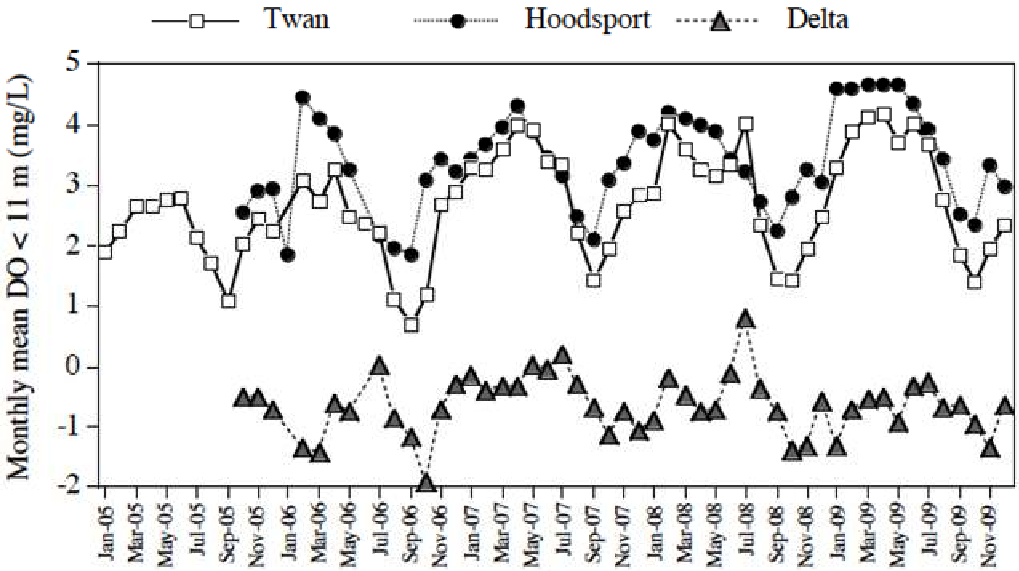
Source: Newton et al. (2011b).

Figure 7: Average dissolved oxygen concentrations in Central Hood Canal (aggregate of all stations, depth > 20 meters)



Source: Warner (2011a)

Figure 8: Seasonal variation in dissolved oxygen at depth at Hoodsport (near the Great Bend) and Twanoh (Lynch Cove)



Source: Brett (pers. comm., 2011c)

Low levels of oxygen result from the complex interaction of physical, chemical, and biological processes. As noted in the introduction, the physical characteristics of Hood Canal (length, depth, and sill-restricted circulation) contribute to a baseline condition of naturally low oxygen levels at depth. Human activities that release nutrients into the Canal can further deplete oxygen concentrations. The relative amount of sources associated with humans, compared with those from natural sources, will define the magnitude of human impact on oxygen concentrations.

Oxygen depletion occurs as a result of organic matter decomposition, and phytoplankton represent a major source of organic matter. Phytoplankton, like other plants, require sunlight and nutrients to grow. These nutrients include both nitrogen and phosphorus. The Redfield ratio is one indicator of the relative nutrient proportions, and phytoplankton require more nitrogen than phosphorus. In most rivers and lakes, the availability of phosphorus in the photic zone controls the growth of phytoplankton. Because of the relative abundance of phosphorus in marine environments, phytoplankton in estuarine and marine waters are generally growth-limited by the supply of nitrogen (primarily nitrate and ammonium).

In the spring and summer, nitrogen concentrations decline in the surface layer as available light increases and phytoplankton consume nutrients. This productivity often depletes nitrogen before phosphorus in the surface layer as phytoplankton consume the available inorganic nitrogen (see Figure 5 for examples of observations of the decline in surface nitrate in Lynch Cove). As the figure indicates, the nitrate concentrations at the surface are substantially lower than those at deeper levels. Differences in density due to variations in salinity and temperature through the water column lead to stratification (increased salinity and density of

water with increased depth), which limits the amount of bottom-water inorganic nitrogen that replenishes the surface layer. In this circumstance, when the natural nitrogen supply is limiting growth, any anthropogenic nitrogen released to the euphotic zone will contribute to an unnatural increase in phytoplankton. HCDOP concluded from productivity tests that nitrogen supply is the critical factor affecting phytoplankton growth in Hood Canal (Newton et al., 2012), which is typical for marine environments.

Paulson et al. (2006) also noted that internal recycling of DIN appears to influence water column concentrations. They reported that DIN concentrations in Lynch Cove at depth (420 µg/L) were significantly higher than concentrations at the entrance of Hood Canal (280 µg/L). They attributed the difference (140 µg/L) to internal recycling within Hood Canal.

Paulson et al. (2006) also examined water quality samples and conducted isotopic analysis of water samples from various locations in Hood Canal and Lynch Cove. USGS found a similar isotopic signature in upper layer organic matter and lower layer nitrate, and they surmised that nutrient-rich, saline bottom water was largely responsible for sustaining the productivity of the phytoplankton in the upper layer.

Circulation

Figure 9 shows a generalized depiction of the two-layer circulation pattern that is common in estuaries. At the boundary between the fresher surface layer and saltier bottom layer, the out-flowing freshwater entrains some of the bottom water, and this transports nitrogen to the surface layer. The data for Hood Canal indicate a similar overall pattern, and this process is noted in the science primer offered on the HCDOP website (Newton, undated).

Figure 10 shows the more complex patterns in parts of Puget Sound from the Strait of Juan de Fuca moving south through Central and South Puget Sound. Consistent with the idealized circulation in Figure 9, freshwater generally moves seaward to the Strait of Juan de Fuca from the inland bays, and marine water moves into the Sound at depth from the sea. The pattern of shallow sills and deeper basins contributes to variable mixing between layers.

Other factors play an important role in mixing. Tidal energy affects mixing in the interior of Hood Canal, entraining nutrients into the surface layer. Tidal mixing may be significant in Lynch Cove, where the mean tidal range (2-3 meters) is a significant fraction of the mean depth (18 meters) (Brett, personal communication, 2012). Wind also plays a role in mixing and entrainment in the interior bays. Finally, circulation in Hood Canal shows strong seasonal variability, including an intrusion of denser, low-oxygen marine water at depth in August and September (see Figure 11). This variability strongly influences the oxygen distribution.

Figure 9. Simplified two-layer estuarine circulation.

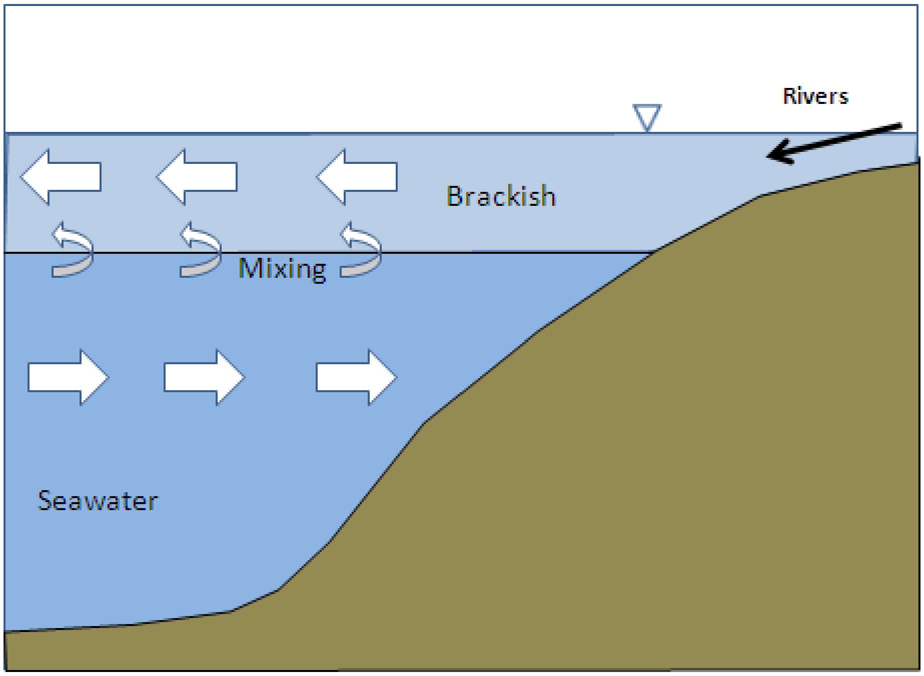
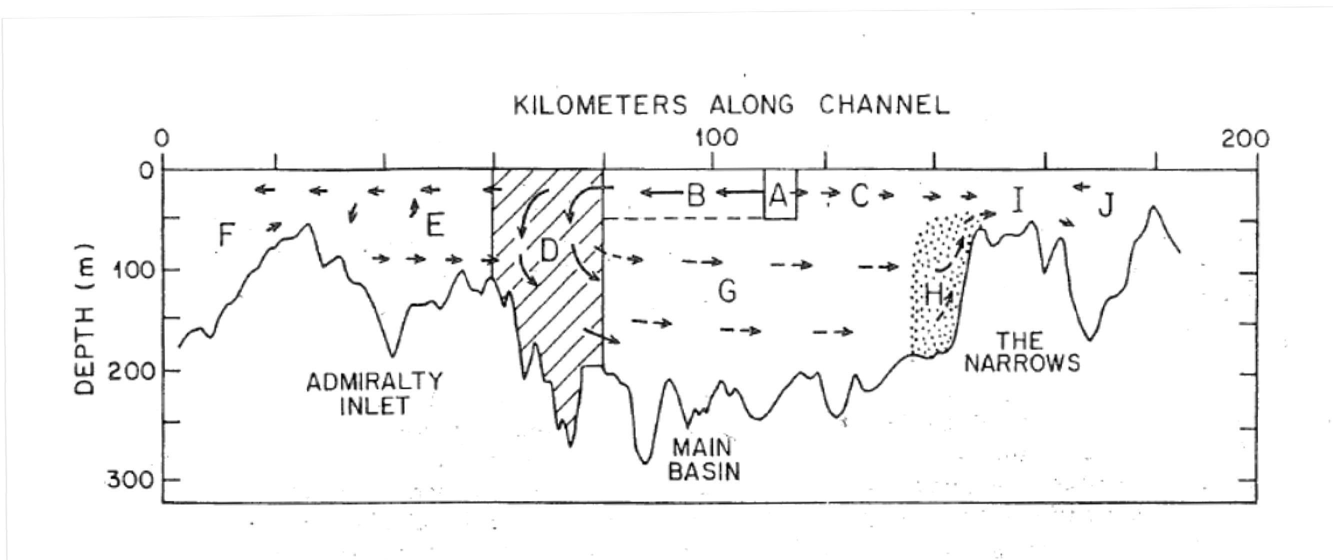
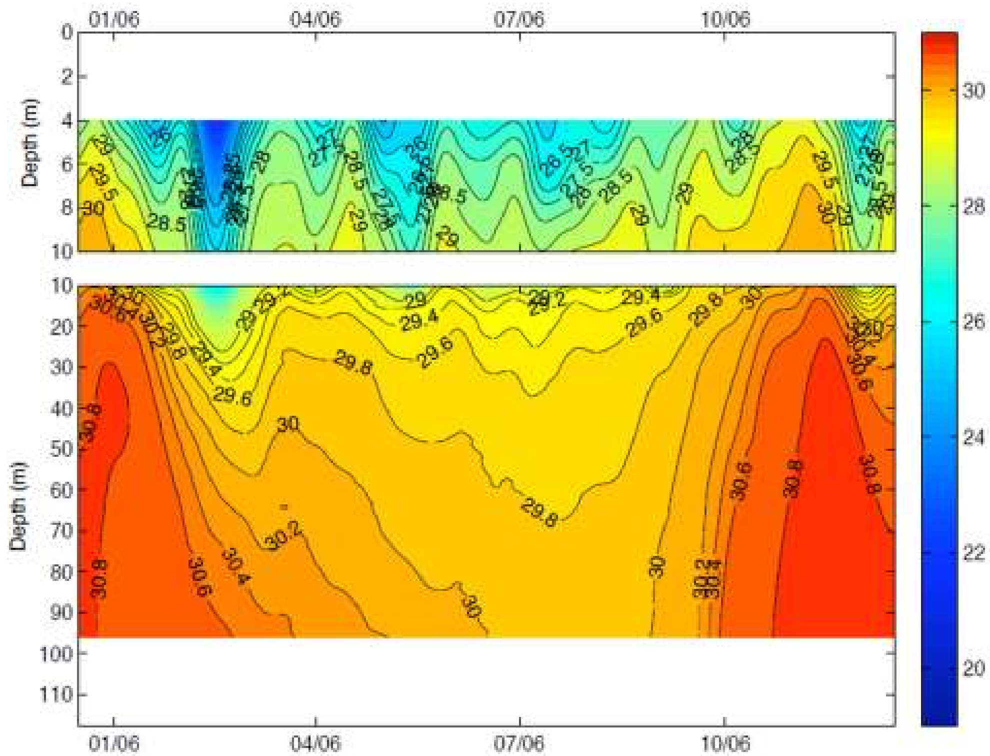


Figure 10. Idealized Puget Sound circulation (letters refer to regions in the source publication)



Source: Ebbesmeyer et al. (1984)

Figure 11: Observed seasonal variation of salinity with depth at Hoodsport. Fall ocean water intrusion appears as higher-density (psu) water.



Source: Kawase and Bahng 2010

HCDOP researchers also identified a previously unrecognized circulation pattern in the Hoodsport area. Based on current velocity measurements recorded near the Hoodsport ORCA buoy, Mickett et al. (2011) identified a subsurface seaward outflow near the middle of the channel that appears in August/September and coincides with periods of low dissolved oxygen. The source of this low-dissolved oxygen water has not been isolated; the data used to identify the phenomenon bound the source as less than 50 meter water depths landward of Hoodsport but cannot resolve the source further. This seaward outflow adds complexity to circulation patterns around Hoodsport and is discussed further in Question 3.

While this review focuses on potential impacts associated with human-caused nitrogen releases, human development can also alter circulation patterns in an estuary or fjord. Circulation, in turn, can alter dissolved oxygen directly (e.g., by changing advective transport of oxygenated water) or indirectly (e.g., by altering biological processes that affect dissolved oxygen). These circulation influences are discussed only briefly because they have not been evaluated quantitatively (see discussion of factors that have not been explored to date in the Uncertainty chapter).

Fish Kills

While low dissolved oxygen is a concern in many areas of Puget Sound, Hood Canal is a primary location of interest because of multiple fish kill events that were attributed to extremely low dissolved oxygen levels. Fish kills have been associated with wind events that align with the axis of the Canal and lead to upwelling of hypoxic bottom waters into shallower depths where biota are not accustomed to such low oxygen levels (Newton et al., 2011c; Kawase and Bahng 2011; Kawase 2007). The fish kills have generally occurred in September on the southwestern shore of Central Hood Canal (from Lilliwaup to Potlatch), which is substantially deeper than Lynch Cove.

HCDOP researchers successfully identified the mechanisms leading to fish kills. Dense marine water enters Hood Canal and lifts water with low oxygen levels up to just below the water surface. As river inflows decline during the dry season, this freshwater cap floating on the surface thins. Southwest wind events push this thin cap to the north, which allows low-oxygen water beneath it to surface rapidly. In a matter of hours, oxygen levels rapidly decrease in the sensitive nearshore regions of southwestern Hood Canal, which causes the fish kills (see Newton et al., 2011c and Kawase and Bahng 2011).

The proximate cause of the fish kills is a natural event initiated by wind. The human influence on fish kills is represented by how much humans decrease already-low oxygen levels in various parts of Hood Canal. Later in this report (see Question 3), we summarize and interpret the available estimates for these human impacts.

Review Considerations

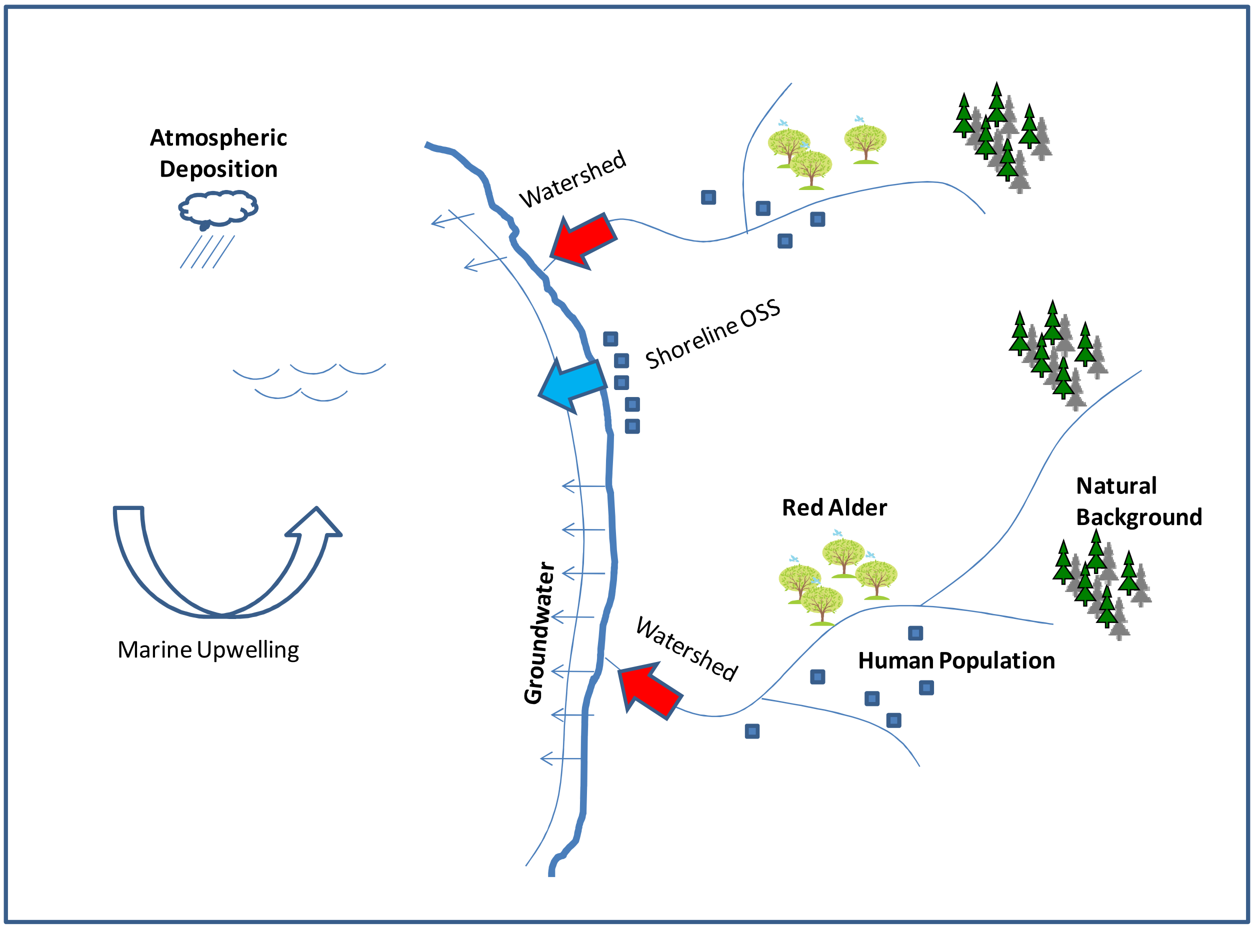
With this background, we now pose the three questions related to human impacts on hypoxia in Hood Canal, provide answers based on available information, and describe the uncertainties around those answers. The sections below balance the need to communicate key components of a complicated system with the need to remain concise. Each source of information mentioned briefly in this document delves into far greater detail than represented in this summary document.

This report includes both qualitative discussions of dominant processes and quantitative analyses. Numeric estimates are expressed as single values or a range in summary tables. These estimates are "best estimates" from a particular analysis. Substantial uncertainty surrounds many of these estimates. The sections below provide qualitative discussions of key areas of uncertainty related to data limitations, system complexity, and analytical methodology. We have also constructed a quantitative, probabilistic analysis of uncertainty in the estimates of human impact to dissolved oxygen in Lynch Cove from simple aggregated models. Uncertainty is discussed within the three questions below and in a separate section. .

Question 1: How much nitrogen do humans and other sources in the watershed contribute to Hood Canal?

This first question requires a variety of information sources such as measurements, land cover, models, literature, and assumptions to estimate nitrogen releases to Hood Canal from the surrounding watershed. Subsections summarize inputs from a wastewater treatment plant (Alderbrook Resort), tributaries (watershed), groundwater and shoreline onsite septic systems (OSS), and atmospheric deposition (see Figure 12).

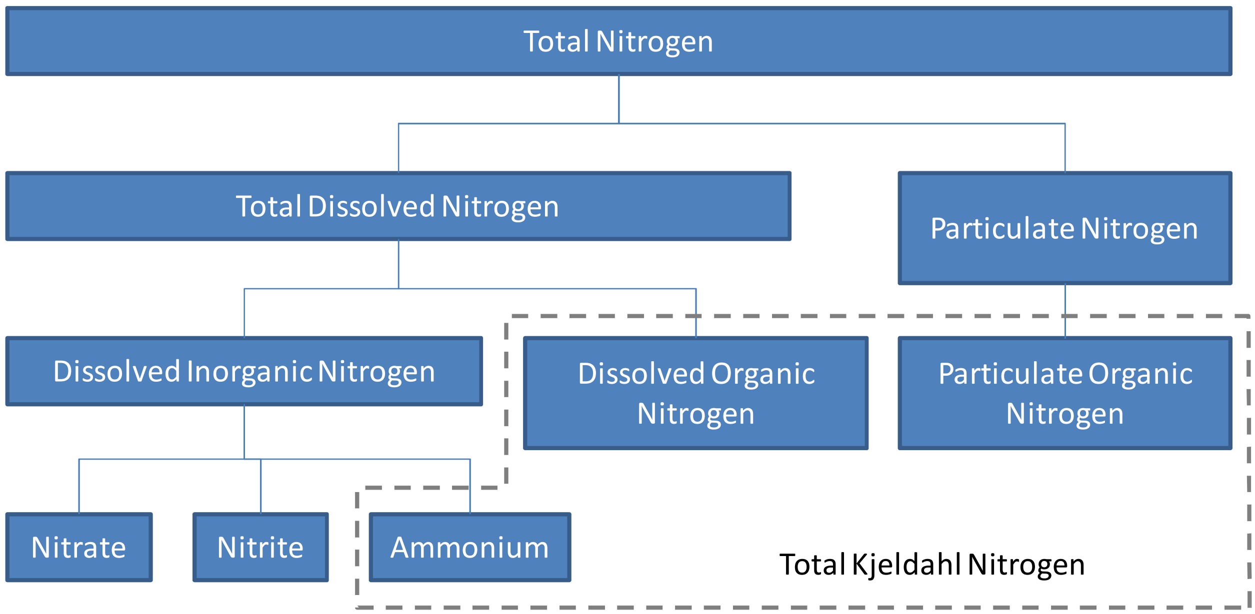
Figure 12: General Nitrogen Sources to Hood Canal



Measured Forms of Nitrogen

Researchers estimate Hood Canal nitrogen loading in a variety of forms of nitrogen. Figure 13 presents the definitions and relationships among the forms used in different source documents.

Figure 13: Forms of nitrogen and relationships among variables



In this review, we compile nitrogen in units reported in the Hood Canal source documents but focus on DIN as the predominant and biologically available form of nitrogen. Because DIN estimates do not include the dissolved organic fraction of nitrogen, the use of DIN values will lead to lower loading estimates than analysis of TDN. However, best available information suggests that DON is a minor fraction in annual loading (Mohamedali et al., 2011) although it may be more significant in freshwater during individual storm events (Ward et al., 2012).

Alderbrook Resort Wastewater Treatment Plant

The only permitted wastewater discharge to the marine waters of Hood Canal is the Alderbrook Resort. The National Pollutant Discharge Elimination System (NPDES) permit now requires monitoring of nitrogen concentrations (nitrate plus nitrite, ammonium, and total Kjeldahl nitrogen). The plant conducted monthly monitoring from February 2009 through January 2010 then decreased frequency to quarterly monitoring (Ecology 2008).

Paulson et al. (2006) quantified this source as 1 metric ton (MT) per year DIN based on typical, but not plant-specific, wastewater concentrations. Paulson et al. (2006) also cited a previous estimate of 0.17 to 1.8 MT/yr by Fagergren et al. (2004), also based on typical wastewater treatment plant effluent concentrations. No other published information quantified this source of nitrogen to Hood Canal. However, based on data available from the Washington State Permit and Reporting Information System (PARIS), the Alderbrook Resort produces 0.13 MT/yr of nitrogen (0.011 mgd, 8.9 mg/L total nitrogen). This concentration is lower than that produced by large plants in the Puget Sound region but typical of smaller plants that achieve

greater nitrogen reduction as a result of plant capacity or operational factors (Mohamedali et al., 2011).

Total Tributary Loadings (Natural plus Human)

Prior to the creation of the HCDOP, a study of nutrient loadings to Puget Sound was conducted by USGS (Embrey and Inkpen, 1998). While not focused on Hood Canal, this study analyzed monitoring data for the larger tributaries to the Canal, including the Skokomish, Hamma Hamma, Dewatto, Dosewallips, and Duckabush rivers. The dissolved inorganic nitrogen (DIN) concentrations in Hood Canal rivers ranked among the lowest in the Puget Sound tributaries analyzed in the study. The study was focused on total loadings rather than the human-caused fraction of loadings. Embrey and Inkpen (1998) estimate 304 metric tons of total nitrogen per year to Hood Canal from major rivers. Load estimates did not account for the smaller watershed areas adjacent to Hood Canal.

USGS (Paulson et al., 2006) summarized available data between 1959 and 2002 for DIN concentrations in major tributaries to Hood Canal. The median concentrations ranged from 40 µg/L (Duckabush River) to 370 µg/L (Union River). The Skokomish, the largest and most frequently sampled river, contained DIN ranging from non-detectable to 720 µg/L, with a median value of 90 µg/L. Monthly DIN loads for monitored and un-monitored subbasins were calculated using mean streamflow, the distribution of the monthly streamflow over the annual cycle, and the mathematical relationship between DIN concentration and streamflow. The annual DIN load from tributaries to central Hood Canal (south of the Hood Canal bridge) and Lynch Cove (east of Sisters Point) combined was an estimated 493 metric tons per year. Monthly totals are shown in Figure 14 below, along with estimated regional groundwater and atmospheric loads. Paulson et al. (2006) did not estimate the human contribution within the total tributary loadings.

More recently, Steinberg et al. (2010) used two years (2005-2006) of monthly data collected by HCDOP from 43 Hood Canal tributaries; these analyses were also presented in Richey et al. (2010). Steinberg et al. (2010) directly calculated the nitrogen loadings entering Hood Canal from monitoring data at the mouths of tributaries. They also constructed statistical models to estimate the fraction of the measured tributary loadings that is attributable to natural contributions (conifer forest) and human contributions (OSS and other residential activities, some portion of alder forests) in each sampled watershed. The statistical model did not change the total loads, which were calculated from monitoring data, but apportioned the total among different sources as described in the next section.

Figure 14. USGS estimates of monthly DIN loads to Hood Canal.

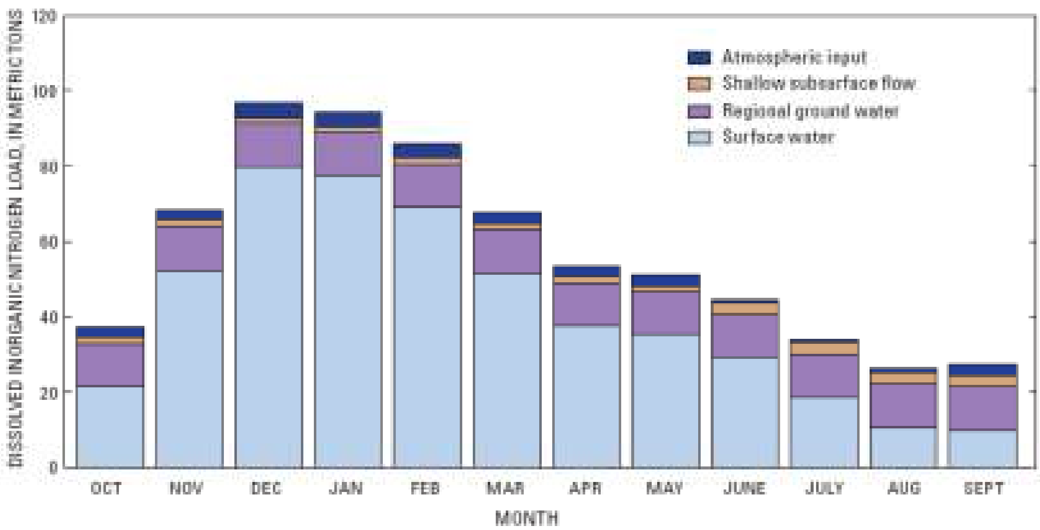
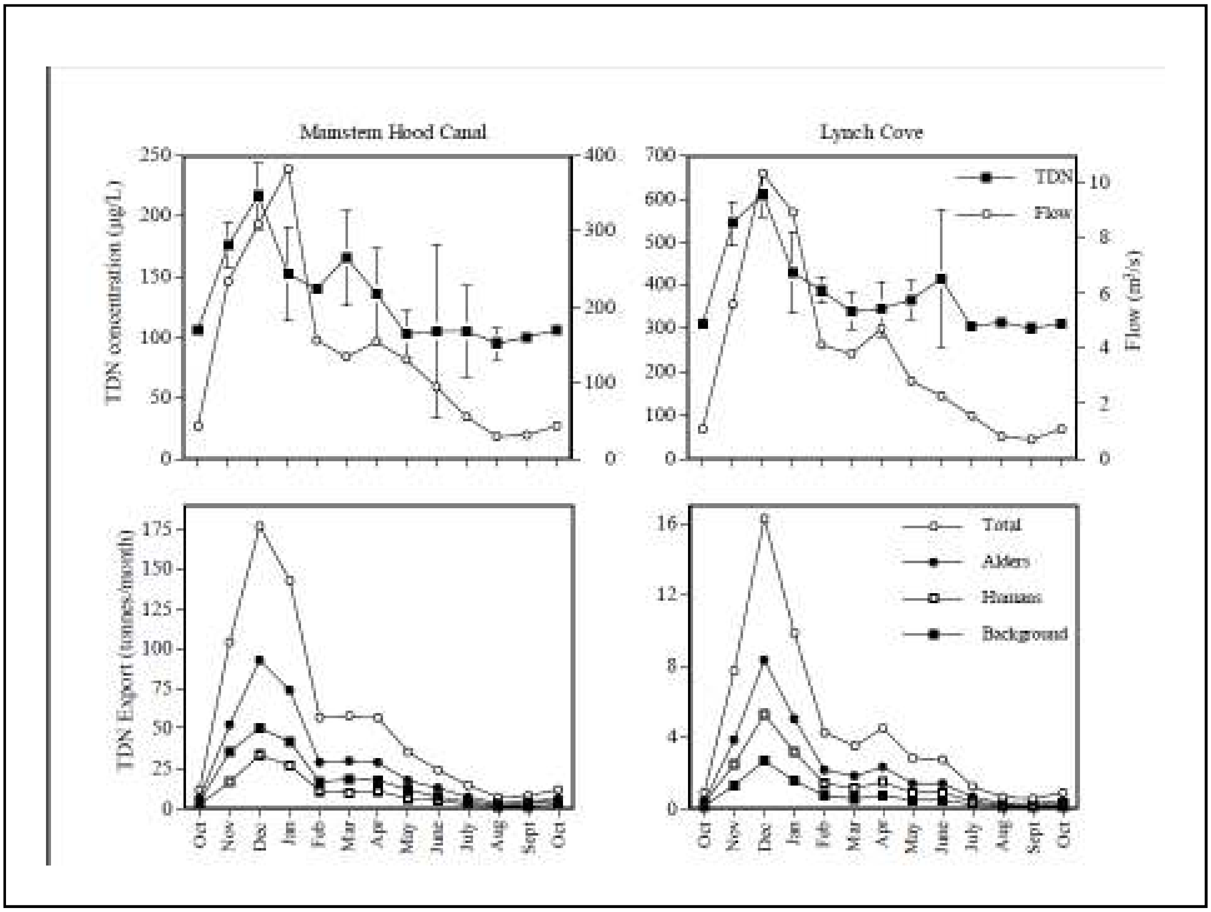


Figure 6. Monthly loadings from four sources of dissolved inorganic nitrogen to the surface waters of Hood Canal, western Washington, 1971-2002

Source: USGS (Paulson et al., 2006)

Steinberg et al. (2010) reported that approximately 700 metric tons/year of total dissolved nitrogen (TDN) enter Hood Canal from tributaries. This is higher than the value in Paulson et al. (2006), which calculated dissolved inorganic nitrogen (DIN) loading. Steinberg et al. (2010) estimated that 21-34% of the total nitrogen from tributaries was dissolved organic nitrogen (147 – 238 metric tons/year). This indicates that the Paulson et al. (2006) estimate of 493 metric tons/year DIN is generally consistent with the later estimates of 700 metric tons/year TDN by Steinberg et al. (2010) and reported in Richey et al. (2010).

Figure 15. Monthly TDN in tributaries to the main arm of Hood Canal and Lynch Cove.



Source: Steinberg et al. (2010) and Richey et al. (2010)

Figure 15 shows the measured flows, concentrations, and loadings from Steinberg et al. (2010) and Richey et al. (2010), as well as the source contributions based on the statistical model. In the upper left panel of the figure, the low nitrogen concentrations in the mainstem (central) Hood Canal align with the conclusion of Embrey and Inkpen (1998) that the rivers along the central Hood Canal carry some of the lowest concentrations in Puget Sound. The figure also shows that TDN concentrations in Lynch Cove tributaries are 2 to 3 times higher than concentrations in tributaries to central Hood Canal and more typical of watersheds with greater development levels (Mohamedali et al., 2011). The greater level of human activity, including OSS, other residential development, and a portion of red alders in the Lynch Cove area has contributed to the increased nitrogen concentrations in local tributaries.

There is significant seasonal variation in nitrogen loading from tributaries. A large fraction of the nitrogen loading to Hood Canal occurs between the months of November and January, when dissolved oxygen concentrations in Hood Canal are recovering from low levels in the late summer. Higher tributary flows and loadings in the fall and winter increase the estuarine circulation, replenishing the area with marine water with higher dissolved oxygen (Brett, 2012).

All of the studies provide credible, measurement-based estimates of the nitrogen entering Hood Canal from tributaries, but the more recent estimates (Steinberg et al., 2010 and Richey et al., 2010) are likely more robust because they are supported by the larger monitoring database for more tributaries collected by HCDOP in the last few years compared with historical compilations based on larger rivers only.

Human Contributions within Tributary Loadings

The statistical model developed in Steinberg et al. (2010) and reported in Richey et al. (2010) apportioned the total tributary loads summarized in the previous section among natural and human sources. Total loadings were reported for Hood Canal, but Lynch Cove loadings were only described as percentages of the totals that are presented in figure form only. From the monthly loading figure for Lynch Cove (see lower right panel of Figure 15), we visually estimate that the total is approximately 60 MT/yr. Table 2 applies the percent contributions among sources to this estimate.

Table 2. Tributary Loads to Hood Canal and Lynch Cove by source (Steinberg et al., 2010; Richey et al., 2010)

Hood Canal

Source	**TDN Loading (MT/yr)**	**Fraction of Total Tributary Loading[3]**
Natural (background)	251	35%
Mixed Forest (Red Alder)	354	51%
Human Population	95	14%
Total	***700***[2]	

.

Lynch Cove

Source	**TDN Loading (MT/yr)**	**Fraction of Total Tributary Loading [3]**
Natural (background)	8	13%
Mixed Forest (Red Alder)	34	56%
Human Population	18	31%
Total	***60***[1,2]	

[1] Estimated by visual inspection of Figure15.
[2] Steinberg et al. (2010) calculated the Figure 15 loads from measurements of flow and concentration at tributary mouths and extrapolated to unsampled catchments using unit-area loadings.
[3]Percent contribution estimated using statistical model

Red alders are a natural component of riparian forests but are more prevalent now than occurred historically (Collins and Montgomery, 2002; Davis, 1973; Brandenberger et al., 2008). Red alder root nodules fix atmospheric nitrogen and produce leaves with higher nitrogen than other plant species (Roberts and Bilby, 2009). Nitrogen leaches from the leaves during rain events. Leaves and other plant materials fall to the ground seasonally, and decomposition processes release nitrogen from atmospheric sources to the soil and water in red alder forests. Roberts and Bilby (2009) concluded that red alder contributed to a 54% increase in nitrogen delivery to small streams (Roberts and Bilby, 2009), and Volk (2004) found that this red alder contribution results in higher stream nutrient concentrations. No study prior to Steinberg et al. (2010) estimated red alder contributions to Hood Canal tributaries.

Richey et al. (2010) also employed a mechanistic modeling approach using DSEM, composed of a watershed model (DHSVM) linked to a solute export model (SEM), to derive estimates of tributary loading for comparison to the statistical approach. Richey et al. (2010) reported model results, which indicated a higher natural loading and lower population and red alder contribution compared to the statistical model. However, Richey et al. (2010) provided only a general description of the DSEM model. A manuscript is pending and no details of the model were available for review.

Overview of Methods to Estimate Loadings from Groundwater and Shoreline Onsite Septic Systems

Tributary estimates do not account for all of the inputs of nitrogen to Hood Canal. In particular, nitrogen loading from shoreline areas not drained by tributaries but contributing through groundwater discharging to the marine waters of Hood Canal is not captured in the tributary analyses. Based on water balance calculations and thermal images, a proportion of groundwater in the watershed discharges directly to marine waters, and an unknown fraction discharges to deep aquifers (see thermal images of shoreline plumes in Sheibley et al., 2010).

Shoreline groundwater loading has particular importance in Hood Canal, because treated wastewater is generally discharged to groundwater via onsite sewage systems (OSS) from developed properties. The vast majority of OSS in Hood Canal serve residential properties. Upland OSS loadings enter Hood Canal as part of the measured nitrogen loadings at the tributary mouths. Tributary measurements, however, do not capture the flux of nitrogen from shoreline OSS, and a significant portion of the population is located in close proximity to the shoreline.

We used two methods to estimate shoreline groundwater contributions: (1) loading calculations using monitoring data from shoreline seeps and groundwater discharge estimates, and (2) per capita loading estimates using population data along the shoreline outside of the tributary monitoring areas. Estimates from both methods are described in the following sections.

Uncertainty is greater in estimates of nitrogen loading from shoreline groundwater than tributary loading due to sampling limitations and complicated groundwater hydrology. A

commensurate monitoring program to capture all groundwater sources would require subsurface sampling along the entire shoreline. While complete sampling is infeasible, an assessment using available field measurements can provide reasonable estimates of the shoreline groundwater loading. Total groundwater flows can be estimated using a water budget developed from tributary flow and precipitation data, and nitrogen concentrations in groundwater can be estimated from field sampling of shallow groundwater and shoreline springs. We analyzed the available flow and nitrogen data to estimate the shoreline nitrogen loading below.

Shoreline OSS loadings have also been estimated by several researchers based on per capita loading calculations, using shoreline population, wastewater nitrogen content, and an assumed loss of nitrogen in groundwater prior to discharge to marine water. Since these OSS loading estimates are based on a population extrapolation and not measured groundwater nitrogen concentrations, it is important to evaluate whether the per capita estimates are consistent with available monitoring information.

Shoreline OSS Loading based on Field Sampling

Groundwater Flow

Paulson et al. (2006) produced the only available estimates of groundwater inflow to Hood Canal and Lynch Cove based on a detailed water budget. After developing tributary flow estimates for both sampled and unsampled watersheds, they calculated the groundwater flow as the difference between annual rainfall (minus evapotranspiration) and tributary flow. The resulting annual average groundwater flow into Hood Canal was 7.3 m^3/sec (258 cfs). Based on the estimated average flow in tributaries of 142 m^3/sec (5,013 cfs), they estimated that groundwater is 5% of the annual average freshwater inflows to Hood Canal.

Paulson et al. (2006) did not explicitly report the Lynch Cove groundwater flow, but they applied the water budget to Lynch Cove watersheds and reported the estimated monthly DIN loading. Based on the loading estimate and assumed DIN concentrations in groundwater (both discussed below), the estimated groundwater flow to Lynch Cove was 1.1 m^3/sec. The analysis assumed that groundwater flow was constant over the year with no seasonal variation. Paulson et al. (2006) did not analyze the distribution of rainwater recharge and groundwater flow between shallow and deep aquifers. In our analysis of human-caused groundwater nitrogen loadings, we have conservatively assumed that the entire groundwater flow enters the euphotic zone of Hood Canal (approximately the top 10 meters of the water column) to evaluate Question 2 below.

Simonds et al. (2008) also estimated groundwater flows for Lynch Cove, but some of the methods resulted in very high flows (e.g., 22 m^3/s for one method) that are not supported by the water budget analysis of Paulson et al. (2006). These high fluxes were also reported in a journal article (Swarzinski et al., 2007) that is no longer supported by best available information. Based on adjustments in methodology and additional field data, USGS researchers indicate that their most recent estimates for the total groundwater discharge to Lynch Cove marine waters range from 0.5 to 1.9 m^3/s (Sheibley, pers. comm., 2011).

Brett (2011d) used a measurement-based nitrogen flux calculation in Lynch Cove as a check on estimates for shoreline OSS loadings. Rather than extrapolating a total water budget for the Lynch Cove watershed as in Paulson et al. (2006), Brett (2011d) used a simpler approach to estimate a residual flow (0.5 m^3/s) based on the annual precipitation (minus evapotranspiration) over the nearshore areas not captured by tributary sampling. This "unsampled flow" could enter Lynch Cove via surface water or shallow groundwater, where relative contributions are unknown. Given that these areas do not have defined streams, much of this flow likely enters Lynch Cove as shallow groundwater.

Shoreline Groundwater Nitrogen Concentrations and Loading

Paulson et al. (2006) estimated "regional groundwater" loading to Hood Canal and Lynch Cove based on nitrogen concentrations in well samples and flow estimates. They found highly variable concentrations, ranging from 60 to 1,000 µg/L DIN, potentially reflecting different levels of human influences along with natural variation. They selected a mid-range value of 600 µg/L DIN to estimate groundwater loading. Coupled with the flow estimate, this led to an estimated DIN loading of 138 MT/year from regional groundwater to Hood Canal. For Lynch Cove, they estimated that regional groundwater contributes a DIN loading of 1.7 MT/month in September and October.

Simonds et al. (2008) estimated total groundwater loading to Lynch Cove using measured nitrogen concentrations in wells, springs, and piezometer samples. The average total nitrogen concentration in 23 samples was reported as 330 µg/L, and the average nitrate concentration was reported as 310 µg/L. We found apparent errors in the reported means based on the raw data provided in Simonds et al. (2008), computing an average total nitrogen concentration of 220 µg/L and nitrate concentration of 200 µg/L. Assuming negligible particulate nitrogen in groundwater, the small difference in these two values (20 µg/L or 9% of the TN) is attributed to ammonium and dissolved organic nitrogen. We used 220 µg/L to represent the Simonds et al. (2008) estimates because it accounts for all DIN components but also includes dissolved organic nitrogen. Based on this information, it is reasonable to assume the organic fraction of nitrogen in groundwater is negligible and DIN is equivalent to TDN.

For nitrogen concentrations, Brett (2011d) cited the previous compilation of sampling information by Paulson et al. (2006) and noted similar data compilations by Atieh et al. (2008) and Kitsap County (Banigan, 2008). From these sources, Brett (2011d) reported that the average groundwater DIN concentration in drinking water wells and unsampled areas appeared to fall in the range of 500-600 µg/L.

Richey et al. (2010) used the DSEM model to estimate a "shoreline groundwater" loading of 0.5 MT/month to Lynch Cove for the period of June through September. The loads are presented as the sum of 0.2 MT/month from conifer, 0.2 MT/month from alder, and 0.1 MT/month from OSS. The report does not describe how these values were estimated or how

they represent groundwater discharge to marine waters. As noted earlier, the DSEM model has not been fully documented, and no explanation of these values have been published.

Mason County has developed the best available sampling data for nearshore groundwater discharges to southern Hood Canal (Georgeson et al. 2008). Mason County has monitored seeps along the southern shoreline of Lynch Cove and the Great Bend since 2008, including sampling of nitrogen in groundwater seeps, springs, small streams, and bulkhead discharges (see Figure 16). The monitoring program captured all seasons of the year.

Kitsap County also collected near-shore samples along the east shorelines of Hood Canal (Banigan, 2008). Salinity information is not available for the Kitsap County data, which is needed to distinguish freshwater discharges from marine or brackish water pushed into shoreline soils by the tides. Brackish or marine samples could dilute the freshwater nitrogen contributions. Therefore, these data have not been included in these estimates of freshwater discharges.

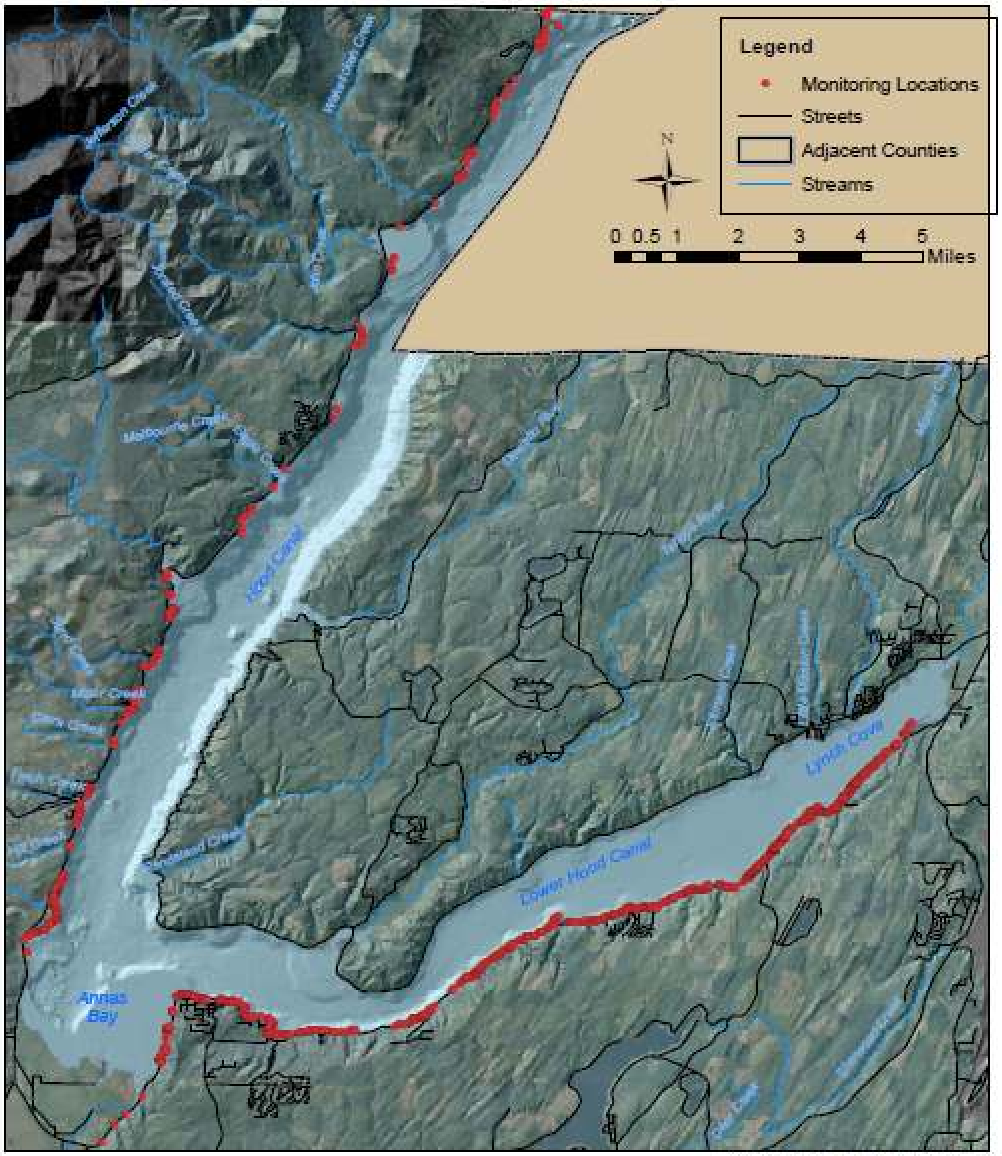
James (pers. comm., 2011a, 2011b, 2012) obtained the Mason County data and analyzed only low-salinity samples (562 samples with salinity <1 ppt) to remove the influence of marine nitrogen. This subset of Mason County samples has a distribution of DIN concentrations that is skewed by a small number of very high concentrations, and James (2011b) recommended use of the median value to represent the central tendency of the data. The median DIN concentration for this subset of Mason County samples was 120 µg/L.

USGS (Sheibley and Paulson, pers. comm., 2011) also analyzed the Mason County data, but they narrowed the dataset to include only seep samples with low salinity levels. They selected this subset of data based on concerns that the full data set included numerous samples from small streams and bulkhead conduits discharging water from unknown sources. The seep samples offer a more representative estimate of the shallow groundwater nitrogen concentrations adjacent to Hood Canal and Lynch Cove. Like James (2011b), Sheibley and Paulson (2011) noted the effect of high outliers in the data and recommended use of the median value for the loading analysis. The median DIN concentration for all of the 382 seep samples in Hood Canal was 250 µg/L. The median DIN concentration was also 250 µg/L in Lynch Cove, where there are significantly fewer seeps due to the mudflat terrain (25 samples, with none along the eastern shore at Belfair).

We reviewed the Sheibley and Paulson (2011) dataset and noted that some seeps, particularly high DIN seeps, were sampled multiple times by Mason County to determine if corrective action on failing OSS had a notable effect on shoreline seep concentrations (Georgeson et al., 2008). Some of the high DIN seeps showed substantial reductions in samples following OSS repair. For example, the seep with highest value (65,000 µg/L in the summer) was re-sampled the following summer, and the DIN concentration had dropped to 250 µg/L (Georgeson et al., 2008). To reduce the bias from multiple sampling of seeps, we reviewed the Sheibley and Paulson (2011) dataset and reduced the distribution to a single value per seep with preference given to the most recent summer sample value. This reduced the number of values from 382 to 325. The median was almost unchanged (250 to 230 µg/L), but the mean dropped from 844 µg/L to 650 µg/L, illustrating the strong influence of the few high values on the mean.

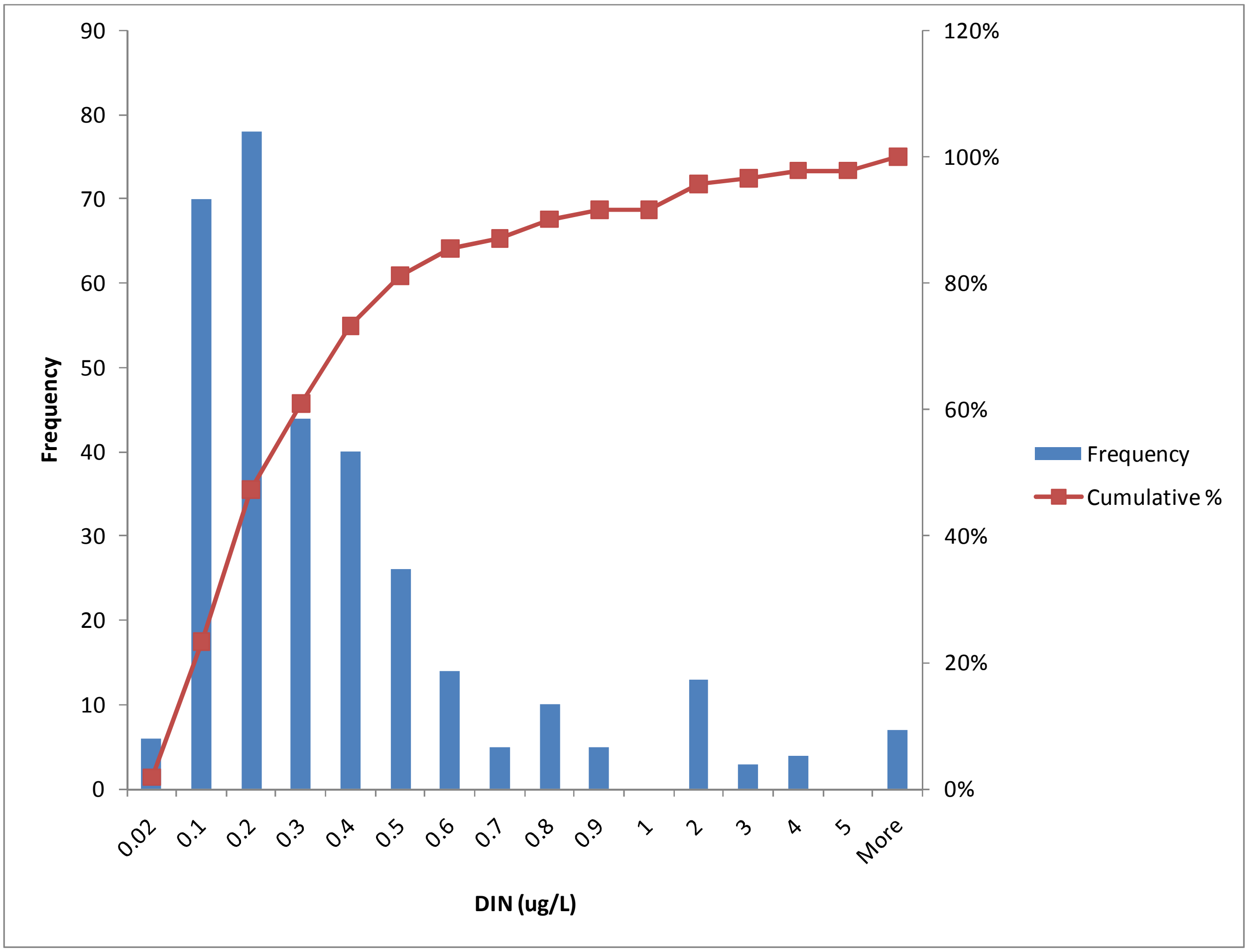
The sampling locations are shown in Figure 16, and the distribution of measured groundwater concentrations samples is shown in Figure 17.

Figure 16 : Mason County shoreline sampling locations



Source: Georgeson et al.(2008)

Figure 17: DIN concentration in freshwater seep samples along Mason County shoreline areas of Hood Canal



Source: Sheibley, pers. comm. (2011), reduced to one value per seep (325 samples) by EPA/Ecology.

Table 3 shows the available groundwater flow and DIN concentration estimates for Hood Canal and Lynch Cove. None of the available studies provide an estimate of natural concentrations of nitrogen in groundwater.

Table 3. Available estimates for flow, nitrogen concentration, and nitrogen loading for groundwater discharges to marine waters

Waterbody	Period	Study/Analysis	Flow (m^3/sec)	DIN (µg/L)	DIN load
					MT/yr
Hood Canal	Annual	Paulson et al (2006)	7.3	600	138
					MT/mo
Lynch Cove	Sept – Oct	Paulson et al (2006)	1.1[1]	600	1.7
Lynch Cove	June – September	Brett (2011d)	0.5[2]	500 - 600	0.7 – 0.8[1]
Lynch Cove (Mason County shore)	NA	Sheibley and Paulson (pers. comm., 2011)	0.5 – 1.9[3]	230[4]	0.3 – 1.3[1]
Lynch Cove (Mason County shore)	NA	James (pers. comm., 2012)	NA	120[6]	NA
Lynch Cove	June – September	Richey et al. (2010)	NA	NA	0.5[5]
Lynch Cove	NA	Simonds et al. (2008)	NA[7]	220[8]	NA

[1] This value was not reported in the referenced document but is readily calculated using the other two values in this row.

[2] Estimated flow from unsampled watershed area in analysis of Steinberg et al. (2010)

[3] No documentation beyond Sheibley (pers. comm.) available for these values at this time.

[4] Median of 325 unique seep samples from Mason County shoreline. The median is 190 µg/L for 21 unique seep samples in Lynch Cove.

[5] Based on DSEM modeling analysis with no documentation at this time.

[6] Median of all Mason county low-salinity data for Hood Canal/Lynch Cove shoreline

[7] Flow values under subsequent review by USGS as they are not bounded by a water balance analysis as conducted in Paulson et al. (2006).

[8] Mean TN value in same sample set. These are corrected values (see discussion of error in text).

The independent review panel recommended use of the mean DIN concentration rather than the median value in the loading calculations (PSI, 2012). As noted earlier, other researchers (James, 2011; Sheibley and Paulson, 2011) recommended use of the median DIN concentration. Because of the influence of a few high DIN samples, the difference between median and mean concentration in the groundwater samples is significant. For the seep samples in Mason County (Sheibley and Paulson, 2011, reduced to single values per seep), the median DIN was 230 µg/L and the mean was 650 µg/L. For 21 Lynch Cove samples, the median was 190 µg/L and the mean was over 2,700 µg/L due to a small number of high sample concentrations.

Flow volumes associated with failing OSS are likely low in comparison to regional groundwater. Preferential flow paths due to groundwater hydrogeology likely produce higher flows and lower concentrations indicative of regional groundwater compared with seeps of effluent from failing OSS (Brett, 2012a). Therefore, the mean is not necessarily more indicative of total loading without concomitant spatially varying groundwater flows and would bias load estimates high.

Rather than selecting a single value to present loadings, we incorporated the entire seep DIN variability into a Monte Carlo uncertainty analysis that is described under Question 3. For summary tables of watershed loading, we present the median and mean DIN concentrations from the distribution to provide a range of loading estimates. This approach was recommended by Brett (2012) in response to the independent panel report. For the lower bound of the loading range in Table 4, we combine the median DIN concentration with the low-end groundwater flow estimate documented in Brett (2011d). For the upper bound, we combine the mean DIN concentration with the high-end flow estimated in Paulson et al. (2006). The high-end flow estimate of Sheibley and Paulson (pers. comm., 2011) was not used because documentation for this estimate is not available.

Table 4: Best estimates of range of seep DIN concentrations, groundwater flow, and shoreline groundwater loadings for Lynch Cove

	Lower Bound	Upper Bound
Shoreline Seep DIN (µg/L)[1]	230	650
Shoreline Groundwater Flow[2]	0.5	1.1
Shoreline Groundwater DIN Loading (MT/mo)	0.3	1.9

[1] Sheibley and Paulson (2011) analysis of data from Georgeson et al. (2008). Dataset reduced to one value per seep by EPA/Ecology. Lower and upper bound are median and mean sample values, respectively.
[2] Lower and upper bound from Brett (2011d) and Paulson et al. (2006), respectively.

Shoreline OSS Loadings based on Per Capita Calculations

In addition to shoreline loading estimates based on seep samples and groundwater flow estimates, three researcher groups (Paulson et al. (2006), Steinberg et al. (2010), and Richey et al. (2010)) estimated nitrogen loadings from shoreline OSS systems using per capita calculations. These studies employed a range of estimates for population, per capita nitrogen loading, and loss of nitrogen in soils between OSS discharge and the Hood Canal water column.

Shoreline OSS loadings to Hood Canal

Paulson et al. (2006) estimated the annual loading from shoreline OSS to Hood Canal and Lynch Cove. The analysis defined shoreline OSS contributions as all homes located within 150 meters of the shoreline. Using 2000 census data and aerial photographs, they estimated an October through May population in Hood Canal of 6,400. For the number of residences (4,900), this equated to an occupancy of 1.3 persons per house. This low occupancy represented the offseason population in Hood Canal, where many vacation properties are not continuously occupied. To account for higher seasonal population, Paulson et al. (2006) used the number of housing units and a summer occupancy rate of 2.5 people/housing unit to obtain a June through September population estimate of 12,200. We found a minor error in this part of the report. The text indicated that an occupancy assumption of 2.2 people per house was used in the calculation, but the actual value used in the calculation was 2.5. A value of 2.2 was clearly the intended value, and this was the value used in the population estimates for Lynch Cove (described below).

Paulson et al. (2006) then used literature values for per capita flow (60 gallons/day) and total dissolved nitrogen concentration from septic systems, assumed the organic fraction is 25% and is removed in soils, and finally assumed a 10% denitrification rate. This resulted in a per capita loading that reaches Hood Canal of 2.95 kg/year, and an annual loading of dissolved inorganic nitrogen to Hood Canal from the shoreline population of 26 metric tons per year. This loading was expressed as DIN but also represented the TDN loading, because it was assumed that none of the organic fraction reached Hood Canal.

Steinberg et al. (2010) did not estimate loadings from shoreline OSSs, but they presented a hypothetical, "worst-case" calculation as an upper bound to make the point that OSS releases would be insignificant compared to marine nitrogen fluxes for the entirety of Hood Canal. They assumed a population of 45,300 for the entire watershed (not only the shoreline) and a per capita nitrogen release to the Canal of 4 to 5 kg per year. They reported that the resulting load (not reported but readily calculated as 204 MT/yr) constituted 0.5% of the marine load for the entire Hood Canal.

Shoreline OSS loadings to Lynch Cove

Using the assumptions described above, Paulson et al. (2006) calculated an average September/October loading of 0.8 MT/month to Lynch Cove from shoreline OSS. The

September contribution was calculated with a higher summer occupancy rate and population (2.2 persons per house, 4,250 people total). This equates to a monthly loading of 1.0 MT/mo for the estimated summer population, higher than for the combined September and October time period.

Paulson et al. (2006) reported shoreline OSS loads as a distinct loading to Lynch Cove, independent of the 1.7 MT/mo loading from "regional groundwater" discussed previously. The rationale was that "regional groundwater" estimates were based on DIN concentrations observed in wells upland from the shoreline, and DIN concentrations in the upland wells are not influenced by shoreline septic influences. Thus, the Paulson et al. (2006) analysis assumed that any impact from shoreline OSS would be seen downstream of these well samples (Sheibley, pers. comm.). This work pre-dated the sampling program by Mason and Kitsap counties, which provided direct measurement of groundwater nitrogen concentrations at the shoreline.

Steinberg et al. (2010) estimated the annual shoreline OSS loading to Lynch Cove and the Great Bend. They estimated a "worst-case scenario" as a population of 4,500-5,000 in the area outside sampled watersheds. Using the same per capita nitrogen value as other studies (4-5 kg per year), they calculated a TDN loading from OSS to Lynch Cove/Great Bend of 21 MT/year, equivalent to 1.8 MT/mo. The population estimate was based on analysis of both residential parcel and census block data (Brett, pers. comm., 2011e).

Richey et al. (2010) considered the USGS (Paulson et al., 2006) study and used a similar calculation method to estimate shoreline OSS loading. They focused on Lynch Cove and examined the June through September period only. Richey et al. (2010) offered 4 new shoreline OSS estimates based on different combinations of population and per capita loading, alongside the Paulson et al. (2006) estimate of 0.8 MT/mo. Per capita nitrogen loading estimates were based on published values of 4.5 kg/year or 2.95 kg/year (4.5 kg/year with 35% nitrogen loss rate in subsurface as used by Paulson et al. (2006); see above)[1]. The estimates for monthly loading ranged from 1.3 to 3.9 metric tons per month to Lynch Cove in the summer. The mid-range load (2.6 MT/mo calculated using a shoreline population of 10,505 people and 2.95 kg per person per year) forms the basis of several subsequent calculations of human contributions to dissolved oxygen decreases. In their report and subsequent discussions, Richey et al. (2010) have emphasized that their estimates were highly uncertain and additional analysis was warranted.

Richey et al. (2010) used significantly higher population estimates than the other researchers. Their report does not identify the buffer width used to estimate shoreline houses and population; however, this was subsequently confirmed to be 1,000 meters (Richey, pers. comm., 2011), which is much larger than the 150-meter buffer used by Paulson et al. (2006). This buffer width incorporates many upland OSS in tributary catchments that are already captured in the

[1] Richey et al. (2010) incorrectly reported the basis for the per capita loading rate used by Paulson et al. (2006). This rate (2.95 kg per person per year) was calculated by reducing the per capita source rate at the residence (4.5 kg per person per year) by 35% based on assumed removal of nitrogen in soils. Richey et al. (2010) listed the Paulson et al. (2006) value but asserted that "other evidence suggests that a more accurate loading rate is 4.5 kg/person-day," in fact, the Paulson et al. (2006) rate was based on 4.5 kg/person-day but incorporated a loss term. This misinterpretation created confusion in the comparisons between loading estimates in Richey et al. (2010). For example, they stated that "all of these calculations assume that the total N loading is transferred directly to the Canal, with no loss", when three of the calculations utilize the Paulson et al. (2006) loading rate (2.95 kg/person-day) that assumes a 35% loss of the per capita nitrogen loading in soils.

tributary loading calculations. As a result, the methodology double-counts a significant number of OSS as both shoreline and tributary sources.

In addition to the buffer width selection, Richey et al. (2010) employed other assumptions that increased the population. For example, they assumed that the July/August population is twice the September/October population in an adjustment to the estimate by Paulson et al. (2006). Additionally, for their mid-range OSS estimates, Richey et al. (2010) assumed that the population is an average of 7,004 people in June and 14,007 people in July/August. The higher population in July/August translates to a very high occupancy rate (3.5 people per house at all times).

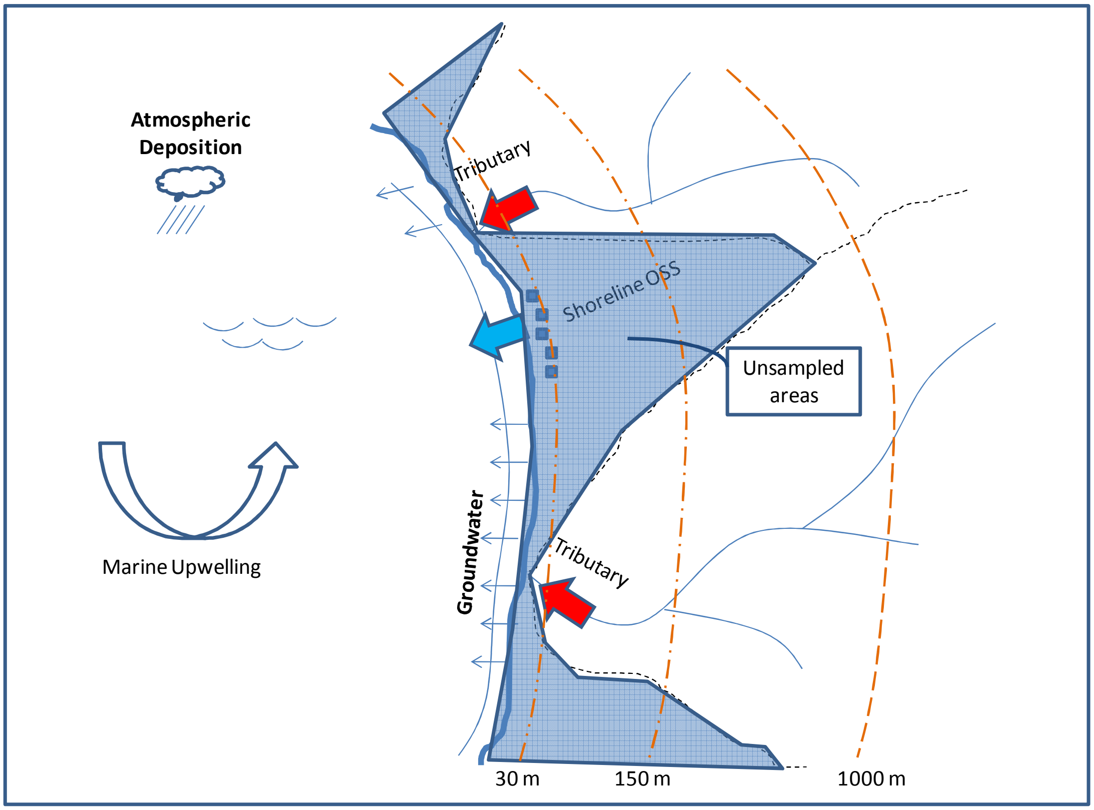
Richey et al. (2010) also calculated a total OSS loading estimate (2.9 MT/mo) to Lynch Cove as the sum of shoreline OSS (2.6 MT/mo), OSS contributions to tributaries (0.2 MT/mo), and "shoreline groundwater" (0.1 MT/mo). This "shoreline groundwater" value is derived from the DSEM model; as noted above, it is not clear how the DSEM loading estimates were derived in terms of the assumed groundwater characteristics and whether these estimates include areas that were captured in the tributary estimates.

A more recent analysis (Horowitz and Peterson, pers. comm., 2011) identified 979 shoreline OSS within a 30-meter buffer. Based on the close proximity to the shoreline of these OSS, a number of these drain fields are tidally inundated. As a worst-case estimate, if one were to assume that all of the nitrogen from the 979 near-shore OSS enters Lynch Cove, and use the assumptions of Paulson et al. (2006) for per capita loading without loss in soils (4.5 kg/year TDN) and occupancy (2.2 persons per house), the estimated loading from the residences within 30 meters of shore would be 0.8 MT/month TDN.

The assumptions and results for the three available studies of shoreline OSS loadings to Lynch Cove (Paulson et al., 2006; Steinberg et al., 2010; and Richey et al., 2010) are summarized in Tables 5 and 6. The studies differ markedly in their geographic scope and OSS assumptions. In addition to differences in study area (Hood Canal or Lynch Cove) and the selected time frame (summer or annual), the primary factors contributing to the variation are differences in estimates for population, per capita loading, and subsurface nitrogen loss.

A major contributor to the differences in study results is the assumed distance from shore (buffer width) used in the shoreline population estimation. As noted above, Paulson et al. (2006) used a 150-meter buffer, and Richey et al. (2010) used a 1,000-meter buffer. Steinberg et al. (2010) used the entire watershed as a bounding calculation and also assessed the unsampled areas. These are shown in Figure 18.

Figure 18. Watershed areas defined in terms of buffer distances, unsampled area delineations, and tributary catchments.



The selected buffer width directly affects the estimated number of residences along the shoreline that contribute nitrogen to marine groundwater loading. The selection of 30, 150, and 1,000 meter buffers led to estimates of 979, 1934, and 3952 housing units, respectively. The available studies do not clearly articulate the basis for selecting a particular buffer. In the course of this review, Brett (pers. comm., 2011f) suggested that assessments should differentiate between those residences where (1) the drain field is close enough to the marine boundary that tidal flushing of the drain field occurs, (2) systems that are not tidally influenced but are not far above saturated soils, and (3) drain fields that are several meters above saturated soils. We believe that future estimates would be improved by a sub-categorization of OSS along these lines as an alternative to calculations that employ a single buffer width. These distinctions are important because research conducted on OSS drain fields in the Hood Canal watershed (Atieh et al., 2008) show that in cases where the drain field is several meters above the saturated zone, nitrogen discharges are stored in the soils until the late fall storms saturate the soils and mobilize stored nitrogen (see also Steinberg et al., 2010). Conversely, for tidal influenced drain fields, nitrogen transport from the septic tank to the marine waters can be very rapid (hours to days).

At greater distances inland from shore, an important factor is the degree to which nitrogen levels attenuate in the subsurface, between the OSS and the location where the groundwater surfaces. A study conducted in the Hood Canal watershed (Atieh et al., 2008) and compilations from New England watersheds with similar geology (Valiela et al., 1997; Daley et

al., 2010) indicate a central tendency of 35-50% loss of nitrogen in effluent plumes down gradient from leach fields. Paulson et al. (2006) and the mid-range estimate for Richey et al. (2010) assumed a 35% reduction in total nitrogen in the groundwater prior to release to the Canal.

Table 5. Estimated shoreline OSS nitrogen contributions based on per capita calculations.

Study	**Annual Shoreline OSS Loading to Hood Canal (metric tons/year)**	**Seasonal Shoreline OSS Loading to Lynch Cove (metric tons/month)**
Paulson et al. (2006)	26	0.8 (Sept – Oct)
Steinberg et al. (2010)	NA[2]	1.8[1]
Richey et al. (2010)	NA	1.3 - 3.9 (June – Sept)

[1]This study reported annual “conceivable” loading without any attenuation of 21 MT/yr, equivalent to listed value of 1.8 MT/mo if distributed uniformly throughout the year.

[2]This study included hypothetical estimates for OSS loadings from population (45,300) in the entire watershed.

[3]Summer contribution to watershed loading not calculated. Annual sampled watershed loading estimate was 58 MT/yr. On annual basis, shoreline OSS loading of 21 MT/yr was therefore 27% of total watershed loading.

Table 6. Assumptions used in per capita estimation of summer shoreline OSS loading to Lynch Cove

Study	Shoreline Width Buffer (m)	Homes (#)	Occupancy (Persons per home)	Shoreline Buffer Population (Persons) and_ (timeframe)	Per Capita Nitrogen (kg/yr)	Summer Loading to Lynch Cove (MT/month TDN)
Paulson et al. (2006)	150	1,934	2.2	4,250 (Sept)	2.95	1.0[5]
Steinberg et al. (2010)	(entire watershed)	NA	NA	4,500 – 5,000 (June – Sept)	4.5	1.8[1]
Richey et al. (2010)	1000[4]	3,952	2.7	10,505 (June – Sept)	2.95	2.6[3]
Horowitz and Petersen (pers.comm.)[2]	30	979	2.2[2]	2154[2]	4.5[2]	0.8[2]

[1]Study analyzed annual loadings only. Listed value is the annual value (21 MT/yr) divided by 12

[2]This study only estimated parcels within 30-m of the shoreline; the remaining values are derived from that parcel count using best available values for occupancy per housing unit and per capita nitrogen contributions.

[3]Richey et al. (2010) refers to this as the “most plausible” value reported from a range of 1.3 to 3.9 MT/mo.

[4] Richey (pers. comm.)

[5] This value was calculated based on Paulson et al. (2006) assumptions for summer population. Paulson et al. (2006) report a September-October loading (0.8 MT/mo), which is a blend of summer and fall population values.

Synthesis of Shoreline OSS Estimates

We reviewed two approaches to estimate shoreline OSS loadings to Lynch Cove. Using shoreline seep sampling and flow balance estimates, the measurement-based approach resulted in

shoreline loading estimates ranging from 0.3 MT/mo to 1.9 MT/mo. The per capita approach employed by several researchers yielded loadings ranging from 0.8 based on 30- or 150-meter buffers to 2.6 MT/mo (based on a 1,000 meter buffer).

Measurement-based ranges represent the best available estimates of shoreline OSS loading. The per capita loading estimates fall within this range for 30-m or 150-m buffer widths, which are less likely to double-count OSS contained within the tributary loading estimates. These values (1.0 MT/mo and 0.8 MT/mo, derived from Paulson et al. (2006) and Horowitz and Petersen (pers. comm., 2011), respectively) fall near the midpoint (1.1 MT/mo) in the range of measurement-based loadings. Nevertheless, we emphasize that the per capita estimates are not based on measured conditions. For this reason, we rely on the measurement-based approach as the best available and consider the per capita estimates as an additional line of evidence.

Shoreline groundwater loading is an important component in the analysis of human impacts on dissolved oxygen impacts. These impacts are estimated later under Question 3, where additional analysis of uncertainty using the Monte Carlo methodology is presented. The Monte Carlo analysis moves beyond point estimates or low/high bounds and employs random sampling of the seep DIN data to capture the full range of uncertainty in shoreline OSS loadings.

Atmospheric Deposition (including Rainfall)

Wet Deposition

Paulson et al. (2006) observed that prevailing winds along Hood Canal are from the southwest, so nitrogen in rainwater is generally not affected by urban areas. They estimated a DIN loading of 30 MT/year from direct precipitation to the water surface of the entire Hood Canal based on the National Atmospheric Deposition Program's Olympic National Park monitoring station. Paulson also estimated a rainfall DIN loading of 272 MT/year to the entire Hood Canal watershed (water and land). As noted earlier, they estimated a tributary DIN export of 493 MT/year.

Steinberg et al. (2010) noted that nitrogen concentrations in rainwater in the Hood Canal watershed are roughly 20 times lower than concentrations observed in the rainwater in the Eastern United States. At the same time, the estimated concentrations (70 µg/L DIN) are similar to tributary concentrations in the most undeveloped watersheds, suggesting that rainwater is probably a significant component of natural background loadings in tributaries. Steinberg et al. (2010) estimated TDN loadings of 47 MT/year to the surface of central Hood Canal and 2.9 MT/year to the surface of Lynch Cove.

Dry Deposition

Steinberg et al. (2010) analyzed available monitoring data for dry deposition of nitrogen. Based on the distance-weighted average from three regional monitoring stations in western Washington, they estimated loadings of 2.4 MT/year to central Hood Canal and 0.2 MT/year to the surface of Lynch Cove.

Summary of Watershed Loadings

The geographical area and time frame for each estimate are critical factors when interpreting the results of the studies of nitrogen loading from the Hood Canal watershed. Table 7 summarizes the various estimates by location and time of year. Each of the studies provided uncertainty ranges, but these are omitted in this summary. We discuss uncertainty later in this report.

Table 7. Watershed nitrogen loads to Hood Canal and Lynch Cove in available studies.

Study Area (Time Frame)	**Hood Canal (Annual)**	**Hood Canal (Annual)**		**Lynch Cove (Sept – Oct)**	**Lynch Cove (June – Sept)**	**Lynch Cove (June – Sept)**
Reference	Paulson et al. (2006)	Steinberg et al. (2010)		Paulson et al. (2006)	Richey et al. (2010)[2]	Richey et al. (2010)[3]
Data Period	1959-2002	2005-2006		2004	2005-2006	2005-2006
Parameter	DIN	TDN		DIN	TDN	TDN
Units	MT/year	MT/year		MT/month	MT/month	MT/month
Tributaries	493	700		0.9	1.4	1.2
Natural	-	251		-	0.7	0.2
Red Alder[5]	-	354		-	0.5	0.6
Population	-	95		-	0.2[4]	0.4
Shoreline OSS	26	NA[1]		0.8	2.6	-
Atmospheric	30	49		0.1	-	-
Wet	30	47		0.1	-	-
Dry	-	2		-	-	-
Regional Groundwater	138	-		1.7	0.5[4]	-
Point Source	1	-		-	-	-
Total	688	749		3.5	4.5	-

[1] Steinberg et al. (2010) did not include a realistic shoreline OSS estimate but rather estimated a hypothetical "worst case" loading from all OSS using the population of the entire Hood Canal watershed (45,300). The OSS load with these assumptions is 204 MT/yr (29% of the sampled watershed load).

[2]Richey et al. (2010) presents several estimates. This column lists values described as "most plausible" loadings. Tributary loadings are based on DSEM modeling analysis (limited documentation).

[3]Richey et al. (2010) presents several estimates. This column lists seasonal loadings estimates for tributaries from the statistical model documented in Steinberg et al. (2010).

[4]The tributary loading from populated areas (0.2 MT/mo) was attributed to OSS. A fraction of the groundwater loading (1 MT/mo) was also attributed to OSS. Therefore, the total estimated OSS loading was 2.9 MT/mo, comprising loadings from shoreline OSS (2.6 MT/mo) and tributary/groundwater OSS (0.3 MT/mo).

[5]Some portion of alder is associated with past human activities, but the relative contribution between natural alder and human-enhanced alder has not been determined.

Table 8 presents our best professional judgment of the potential range of loadings to Lynch Cove based on the supporting information described in this report. The total tributary values are measured loadings based on HCDOP monitoring date, while the source contributions are based on the statistical model described in Steinberg et al. (2010) and Richey et al. (2010). These source contribution estimates were preferred over the DSEM model values reported in Richey et al. (2010), because the statistical model is more thoroughly documented and peer-reviewed. Groundwater and shoreline OSS estimates are carried forward from the groundwater section. As noted previously, there is substantial uncertainty in each point (single value) estimate, and we return to these uncertainties in a Monte Carlo analysis under Question 3.

The total human contribution includes the human contributions within the tributary loadings (0.4 MT/mo) and the shoreline OSS (0.3 to 1.9 MT/mo), while the natural component of the tributary loadings (0.2 MT/mo) is associated with native conifer forests. Some portion of alder is associated with past human activities, but the relative contribution between natural alder and human-enhanced alder has not been determined. For dissolved oxygen impact analysis, the alder contribution is considered human as a conservative assumption.

Table 8. Potential range of watershed nitrogen loads to Lynch Cove in summer (June through September) based on review and synthesis of available studies.

Source	Study	Low Range	% of Total	High Range	% of Total
		TDN (MT/mo)		TDN (MT/mo)	
Tributaries	Steinberg et al. (2010); seasonal estimates reported in Richey et al. (2010)	1.2	80%	1.2	39%
Natural	Same as above	(0.2)		(0.2)	
Red Alder	Same as above	(0.6)		(0.6)	
Population	Same as above	(0.4)		(0.4)	
Groundwater (includes shoreline OSS)	Synthesis of multiple studies[1]	0.3	20%	1.9	61%
Atmospheric Deposition (Combined Wet and Dry)	Paulson et al. (2006); Steinberg et al. (2010)	< 0.1	<0.1%	< 0.1	<0.1%
Total		1.5	100%	3.1	100%

[1] Groundwater flow from Brett (2011d) and Paulson et al. (2006). Median and mean DIN concentration in shoreline seeps (from Georgeson et al. (2008) and Sheibley and Paulson (pers. comm., 2011)). Per capita estimates from Paulson et al. (2006). Near-shore housing units from Horowitz and Petersen (pers. comm., 2011).

These estimates indicate that human-caused nitrogen loadings from the Lynch Cove watershed are substantial when compared to natural watershed loadings. If we conservatively assume that all of the red alder and groundwater loadings are caused by human activity, then natural sources comprise only 6 to 13% of the summer nitrogen loading entering Lynch Cove from the watershed (0.2 MT/mo of the total watershed loading of 1.5 to 3.1 MT/mo).

The watershed loading estimates are carried into Question 2, which also explores the marine contribution to algae growth in Hood Canal and Lynch Cove. Question 3 then builds from these estimates to quantify the relative impact of human loads to DO decreases.

Question 2: How much nitrogen do humans contribute to the surface layer of Hood Canal compared to marine sources of nitrogen?

The surface layer dynamics in Hood Canal are key to linking nitrogen sources to dissolved oxygen impacts. The nitrogen supply in the surface layer drives phytoplankton growth and strongly affects oxygen conditions in the water column. In Question 1, we analyzed nitrogen entering the surface layer from the watershed. The focus of Question 2 is the relative contribution of this watershed nitrogen loading compared to the natural loading of oceanic nitrogen mixing into the surface layer of Hood Canal.

The nitrate concentration at depth in the Great Bend/Lynch Cove area (400 µg/L) is approximately four times higher than nitrogen concentrations in the Skokomish River (median of 90 µg/L reported in Paulson et al (2006)), confirming the importance of marine nitrogen sources. As noted earlier, the net circulation in the Canal transports deep water landward into the Canal. Mixing processes transport nitrogen and other solutes from the bottom waters into the surface layer of the Canal and the surface layer flows seaward. The transport into the surface layer has both advective (upward) and turbulent mixing (bi-directional) components. The relative strength of the advection (also called "upwelling") and mixing (also called "eddy diffusion") processes varies over space and time, complicating efforts to distinguish between them analytically.

Three studies, Paulson et al. (2006), Steinberg et al. (2010), and Devol et al. (2011a), evaluated the relative contribution of watershed discharges and marine advection to the surface layer. These studies used a variety of methods as described below. A fourth study, Kawase and Bahng (2012), evaluated the advective and diffusive flux of nitrogen to the surface layer using a three-dimensional, dynamic, biogeochemical model.

Estimates based on Observations

USGS (Paulson et al., 2006) described the characteristic two-layer circulation of Hood Canal. Based on current meter measurements described in Noble et al. (2006), they estimated the landward transport of inorganic nitrogen in the deep waters of Hood Canal and Lynch Cove using current meter data. Compared to an estimated 688 metric tons per year flowing into the surface layer from the entire watershed, they estimated that 10,100 to 34,000 metric tons of dissolved inorganic nitrogen (DIN) per year enters the Canal from Admiralty Inlet at depth. They also narrowed the analysis to Lynch Cove in September and October, where the marine flux contributed an estimated 132 metric tons of DIN per month to Lynch Cove, compared to 3.6 metric tons per month entering the surface layer from the watershed.

Devol et al. (2011a) reported a mean sub-tidal velocity in the summer of 0.005 m/s at the Twanoh buoy in Lynch Cove. Devol et al. (2011a) did not report the advective flux of nitrogen based on this current speed. Multiplying the mean velocity by the cross-sectional area below the pycnocline at this location (38,960 m^2 in Devol et al. 2011a), we calculate an advective flux of water of 195 m^3/sec. Using the mean nitrate in the lower layer of 333 µg/L from Devol et al. (2011a), we calculate a nitrate flux of 168 MT/month of nitrate from the deep layer to the surface layer. The DIN value would be slightly higher after accounting for the ammonia component.

If the buoy represents the cross-sectional average current velocity, the calculated transport above could be a best estimate. If the buoy is located where current velocities are highest, then the calculated transport is overestimated. It is unclear whether either assumption is valid based on the available studies. In addition, Noble et al. (2006) presents information indicating the uncertainty in the measured current velocities in that study. This information suggests that measurement error/uncertainty in the current meter data is an important consideration when interpreting the data and calculating fluxes.

Estimates based on Aggregated Models

Two studies estimated the flux of nitrogen from deeper waters to the surface layer of Hood Canal and Lynch Cove using "aggregated" or "box" models (Steinberg et al. 2010, and Devol et al. 2011a). These aggregated models are generalized representations of the system that focus on phytoplankton, nutrient, and oxygen relationships and variation in the vertical dimension only. These relationships are assumed to be spatially consistent throughout the waterbody segment as a whole. The aggregated models were used to estimate nitrogen loadings in a two-step process, beginning with estimation of vertical advective flows to the surface water. These flows were then paired with measured nitrogen concentrations in the water column to calculate nitrogen loading.

Vertical Advection of Marine Water

Prior to recent efforts to estimate nitrogen flux to surface waters in Hood Canal, Babson et al. (2006) developed a box model of Puget Sound circulation that divides the estuary into seven basins, with two vertical layers in each basin. This model has been adapted and shared widely by the Washington Department of Ecology in their ongoing analyses of PCB fate and transport in Puget Sound (Pelletier and Mohamedali, 2009). The model provides a time-series estimate of vertical upwelling from the lower layer to the surface layer of each basin. Hood Canal is represented as two connected basins in the model, with the southern basin representing the area encompassing central Hood Canal (south of the sill) and Lynch Cove. For this area, the model predicts an annual average advective flow from the lower layer (depth greater than 13 meters) to the surface layer of 2496 m^3/sec. Since central Hood Canal and Lynch Cove were combined in a single basin, the model does not provide an estimate for Lynch Cove alone.

Steinberg et al. (2010) used a two-layer mass balance of salinity, also referred to as Knudsen's relation, to estimate the vertical advective flow into the surface layers of Hood Canal and Lynch Cove. This analysis focused on annual average flows. The estimated advective flow in Hood Canal was 3025 m^3/sec, which was generally consistent with the results of the model of Babson et al. (2006). The Lynch Cove vertical advection to the surface layer was 35.8 m^3/sec.

Devol et al. (2011a) employed multiple methods to estimate vertical flow to the surface layer in the summer in Lynch Cove. The methods included the two-layer method employed by Steinberg et al. (2010) but applied to the summer season rather than the entire year (Method A). Method B also represented a Knudsen salt balance, but only for the lower layer. The approach accounted for vertical diffusive mixing in addition to advection. Method C employed a nitrogen mass balance to estimate vertical advection, including estimates for denitrification and primary productivity processes. Method D was a tidally-averaged model with several horizontal boxes

and vertically resolved layers to describe estuarine circulation. The range of advective flow estimated using these methods was 20.2 to 73.2 m^3/sec.

Vertical Advective Flux of Nitrogen in Hood Canal

Steinberg et al. (2010) estimated the relative contribution of watershed and marine nitrogen loadings to the surface layer of Lynch Cove and Hood Canal as a whole. The estimated marine upwelling flows were paired with bottom layer average TDN concentrations to derive an annual average loading to the surface layer, which Steinberg et al. (2010) defined as the top 5 meters in Lynch Cove and 9 meters in the rest of Hood Canal. The TDN loading for central Hood Canal (39,000 metric tons TDN per year) generally agreed with the upper range estimates of Paulson et al. (2006). Steinberg et al. (2010) then compared this marine loading to the loadings for watershed discharges, rainfall, and dry deposition. The annual average contribution of marine upwelling to surface layer TDN loadings was estimated at 98% for central Hood Canal, similar to the upper range of Paulson et al. (2006). Using conservative estimates, they estimated that OSS loadings contribute at most 0.5% of the total nitrogen loading to the surface layer of central Hood Canal.

Vertical Advective Flux of Nitrogen in Lynch Cove (Annual Average)

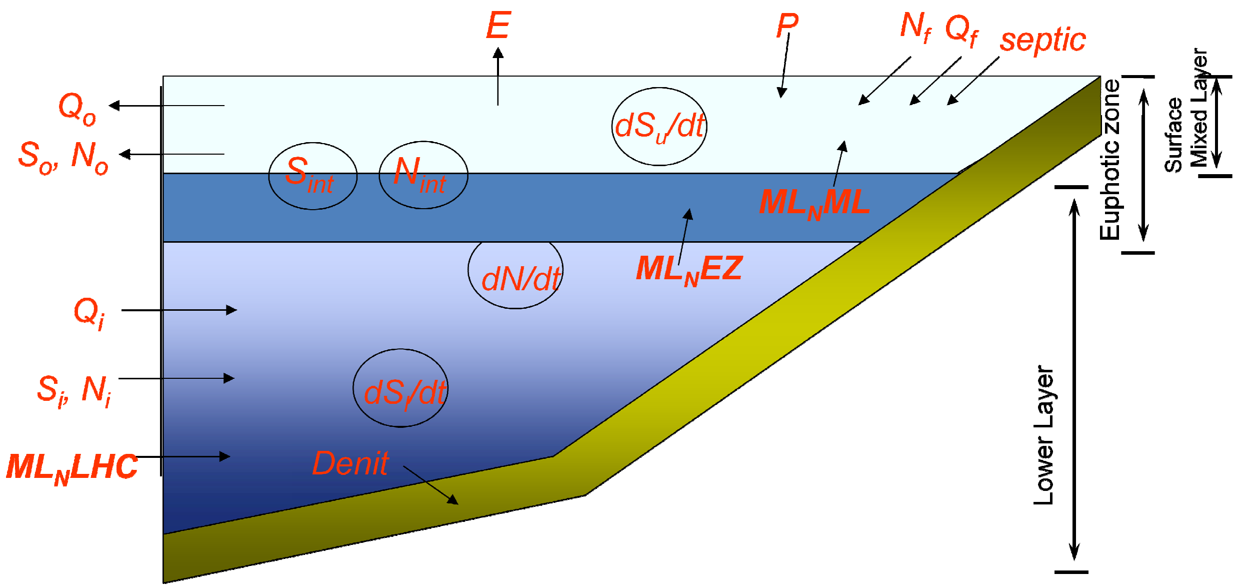
Steinberg et al. (2010) also conducted a separate analysis for Lynch Cove. They estimated a marine upwelling load of 438 MT/year, compared to a watershed load of 55 MT/yr and shoreline OSS load of 21 MT/yr. While they estimated that over 80% of the total nitrogen loading from the tributaries to Lynch Cove was attributable to altered forests and human development, the combined watershed and shoreline OSS loading contributed 15% of the total nitrogen loading to the surface layer of the water column, with marine nitrogen representing the remainder.

Vertical Advective Flux of Nitrogen in Lynch Cove (Seasonal)

Steinberg et al. (2010) did not calculate a seasonal flux of nitrogen into Lynch Cove, but they acknowledged the importance of seasonal factors in the overall comparisons of marine and watershed loading. For example, the summertime contributions of residential nitrogen loadings are likely to be higher-than-average due to the increased summer population. These increased residential contributions occur during a period of diminishing tributary flows and associated red alder influences. In turn, diminished tributary flows result in diminished marine flows required to maintain the salt balance that drives estuarine circulation. These seasonal changes lead to higher shoreline wastewater contributions relative to tributary and marine inputs in the summer months.

Devol et al. (2011a) estimated the relative loading of nitrogen in the summer months from marine and human sources to the surface layer in Lynch Cove. This analysis employed a 3-layer model for Lynch Cove (Figure 19), in contrast to the 2-layer model used by Steinberg et al. (2010. The top layer or “surface mixed layer” is above the pycnocline, which Devol et al. (2011a) define as surface to 6 m. The middle layer (6 to 12 m depth) is the area below the pycnocline but still within the euphotic zone. The lower layer defined by Devol et al. (2011a) (12 m to bottom) is the area of minimal primary productivity and low dissolved oxygen.

Figure 19. Conceptual model used to define layers (see Devol et al. 2011a, for an explanation of terms).



Devol et al. (2011a) analyzed local human and natural marine fluxes of nitrogen to the top two layers as well as the mixing processes that affect oxygen conditions. There are several important assumptions in the 3-layer analysis of Devol et al. (2011a), including:

1. Surface layer algae productivity produces particulate organic matter that settles to the second and third layers. Oxygen production in the surface layer does not transmit oxygen to second or third layers because the pycnocline is an effective barrier to mixing.

2. In mathematical terms, oxygen and nitrogen fluxes are fully balanced in the middle layer, meaning that the productivity associated with natural marine nitrogen sources in the middle layer (due to the vertical marine flux of nitrogen) produces and consumes oxygen in equal measure and does not cause a net depletion of dissolved oxygen in the lower two layers. This means that the only cause of dissolved oxygen depletion in the bottom layer occurs as a result of net organic matter from the surface mixed layer passing through the middle layer to the bottom layer.

   Devol (2012a) noted that, conceptually, some of the surface production is oxidized in the middle layer, and some escapes along with any unoxidized production from the middle layer.

3. In mathematical terms, based on the assumptions above, the net observed oxygen depletion in the third layer is caused entirely by the productivity in the surface layer. Therefore, the fraction of human-to-marine nitrogen loading to the surface layer defines the proportional human impact on the bottom two layers.

   Devol (2012a) noted that some production from each part of the euphotic zone is oxidized in the middle layer by the oxygen produced in the middle layer.

4. Fluxes are based on concentrations at the interfaces between the defined layers instead of layer averages. This results in lower marine fluxes to middle and surface layers than would be calculated using layer averages, because the nitrogen concentrations decrease at shallower depths.

These assumptions have been the subject of debate among Hood Canal researchers. While a 3-layer construct is consistent with observations of phytoplankton productivity below the pycnocline, the application of a 3-layer model to estimate natural and human-caused oxygen impacts is not a commonly used approach. The researchers held a common understanding of the 3-layer construct, captured in a hypothetical diagram (Figure 20) of the natural and impacted condition in Lynch Cove. However, the researchers were unable to reach consensus on the assumptions used by Devol et al. (2011a) despite additional examination (Devol, pers. comm., 2011c). While it was agreed that the nitrogen loading to the surface layer impacts the oxygen concentrations in the bottom layer, there was a difference of opinion on the potential effect of marine nitrogen loading to the middle layer.

Brett (pers. comm., 2011g) offered the following alternative assumptions:

1. Particulate organic carbon (POC) generated by phytoplankton productivity and zooplankton fecal matter in the surface and middle layers have similar settling velocity distributions and mineralization rates.

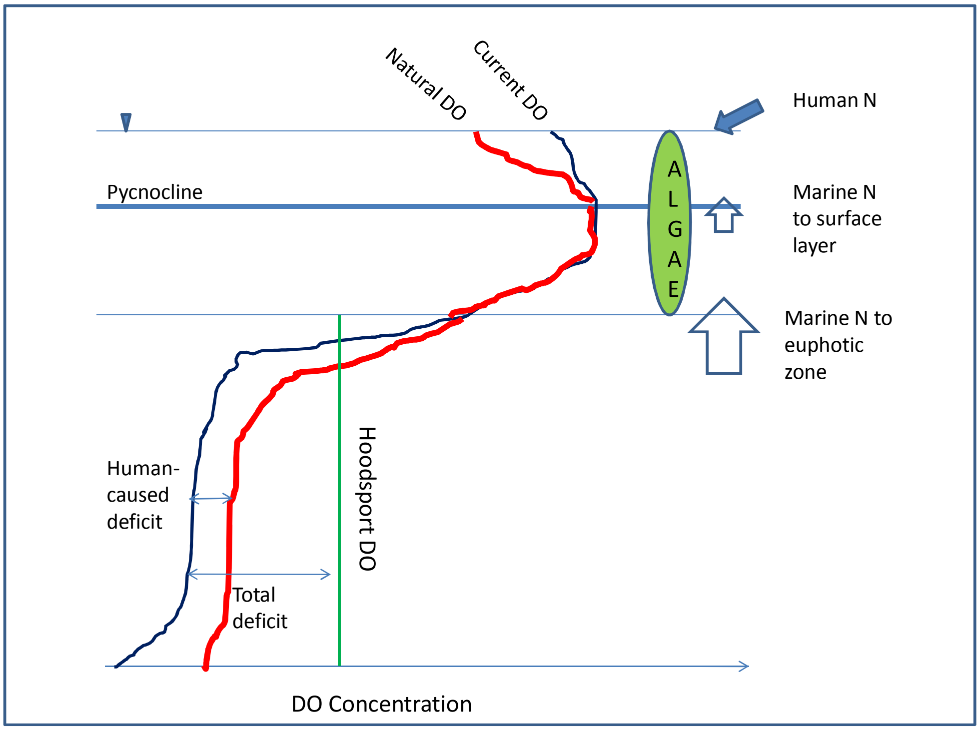
2. POC settling velocity is much higher than the mineralization rate and much higher than the downward diffusion/mixing of dissolved oxygen. Therefore, most decomposition occurs in the bottom layer where dissolved oxygen is also substantially depleted.

3. The fractional mineralization of POC produced within and outside each layer should be considered. For illustration, Brett (pers. comm., 2011g) hypothesized the following: 20% of the POC generated by production in the surface layer is mineralized in the first layer, 20% is mineralized in the middle layer, and 60% is decomposed in the bottom layer. In contrast, 20% of the organic matter generated in the middle layer is mineralized there and 80% is mineralized in the bottom layer.

4. These conditions would produce a surface layer that has dissolved oxygen in equilibrium with the atmosphere (due to gas exchange at the air-water interface), a middle layer that is somewhat supersaturated with dissolved oxygen, and a bottom layer that is very depleted of oxygen.

In contrast to Devol et al. (2011a), these assumptions would not restrict comparisons of human and marine nitrogen loading to the surface layer alone, because the marine loading to the middle layer is assumed to have a net detrimental effect on oxygen in the bottom layer.

Figure 20: Hypothetical Lynch Cove dissolved oxygen profiles for a 3 layer flux analysis



The lack of consensus on the 3-layer model assumptions is important because the estimate of human impact to dissolved oxygen (Question 3, below) is significantly affected by the assumptions of the 3-layer model. Devol et al. (2011a) found that phytoplankton depletes the nitrate concentration at the interface between the surface layer and the middle layer to one fourth of the concentration in the bottom layer. As a result, the marine nitrogen loadings to the middle layer (from the bottom layer) and surface layer (from the middle layer) differ by a factor of four. This difference carries into the oxygen impact calculations in Devol et al. (2011a) and Brett (2010a). As will be discussed under Question 3, Brett (2010a) adapted the 2-layer approach and marine nitrogen loading of Steinberg et al. (2010) in subsequent oxygen depletion calculations for Lynch Cove. Because the top layer of the 2-layer model extends to a depth that is similar to the middle layer depth in the 3-layer model of Devol et al. (2011a), Brett (2010a) used a much higher nitrogen concentration for the marine loading than the surface layer value used in Devol et al. (2011a). This, in turn, led to a higher marine loading and a concomitantly lower relative human impact when compared to the estimates of Devol et al. (2011a). The oxygen impact calculations are discussed under Question 3 below.

The independent review panel identified several fundamental limitations of each of the approaches (PSI, 2012). The panel discouraged the use of Methods B, C, and D and preferred

Method A but only after re-analyzing the layer salinities used to calculate exchanges using the vertical profiles of horizontal velocity and salinity. The panel also identified the lack of tidal dispersion or any other time-varying contribution to salt and nitrate fluxes as flaws in the analyses. They recommended that the uncertainties in the values should be estimated. Finally, the panel did not agree with the approach to estimate the nitrogen flux from the salt balance flux multiplied by a layer or interface concentration. The panel argued that the analysis should account for the horizontal advection of nitrogen in the conservation of mass calculations and pointed to the buoy data to explore the vertical variation in the vertical velocity. No re-analysis has been conducted to address the technical points raised by the independent review panel.

The capabilities and limitations of these aggregated models should be explored further before relying on results for regulatory actions. As noted, these tools are highly simplified representations of a complex, dynamic system. Brett (pers. comm., 2011h) recently noted the complicated vertical structure of chlorophyll-a and nitrogen concentrations in the water column, and this variation raises questions about the simple assumptions of the aggregated models (e.g., depiction in Figure 20 above). For example, vertical plots from different timeframes show phytoplankton peaks and nitrogen depletion occurring both above and below the pycnocline. Devol (2012a) noted that the use of monthly averages of around 300 profiles provide a representative structure of mean condition, recognizing that there is variability around the mean.

Estimates based on ROMS Water Quality Model

The ROMS biogeochemical model of Hood Canal was also used to estimate vertical flux of nitrogen in Lynch Cove in the summer (Kawase and Bahng, 2012). Additional model details are described later in this section. The depths where fluxes were reported varied between Devol et al. (2011a) and Kawase and Bahng (2012). Specifically, the ROMS modeling analysis used a 10 meter depth for the marine flux calculation without referencing the 3-layer construct and 6 meter depth of the surface layer in Devol et al. (2011a)). Kawase and Bahng (2012) did note that nitrogen in the model is depleted by phytoplankton to this depth, deeper than was observed. In response to our inquiries about the comparability of the ROMS and box models, Kawase (pers. comm., 2011e) indicated that the ROMS-estimated marine loadings were most representative of the loadings to the middle layer estimated in the box model analysis of Devol et al. (2011a).

Unlike the steady-state box models, the ROMS model provides a continuous simulation of both advective and diffusive flux for the summer period. Two simulations were conducted. The first was a "climatological" or long-term average condition. The second was a simulation of 2006 conditions. For the period July 1 to September 1, 2006, the average summer flux at 10 meters was 78 MT/mo of nitrogen, composed of 66 MT/mo of nitrate and 12 MT/mo of ammonia. Approximately half the flux was diffusive (34 MT/mo nitrate and 6 MT/mo ammonia). For the climatological scenario, the model predicted a total nitrogen flux of 28 metric MT/mo and did not provide a breakdown of component fractions. There was considerable variability in the magnitude of the flux over time within the summer season for both scenarios.

Kawase (pers. comm., 2011e) noted that the 2006 scenario result for nitrate advective flux (32 MT/mo) was generally consistent with the range of estimates (11 to 59 MT/mo) in Devol et al. (2011a) for advective flux into the middle layer of the 3-layer model. As noted earlier, however, Devol et al. (2011a) did not use the flux to the middle layer from the bottom

layer in the oxygen impact calculations. They used the flux to the surface layer, and there are no ROMS flux estimates for shallower depths to compare to the flux calculated by Devol et al. (2011a).

Summary of Aggregated Model and ROMS Model Estimates of Marine Nitrogen

Table 9 summarizes the various estimates of nitrogen loads to the surface mixed layer or surface plus middle layer (euphotic zone) of Hood Canal as a whole and Lynch Cove as a subset. These include both annual and seasonal values from the box models and ROMS model.

Synthesis of Marine Flux Estimates and Limitations

Issues Identified by Independent Review Panel

The marine flux calculations for Lynch Cove were a focus of the independent review (PSI 2012). The independent review panel raised several issues with the methodology and documentation in the source material, including:

- Method B (lower layer salt balance) was discouraged because it relies on an estimate of eddy diffusivity without corroboration.
- Method C (nitrogen mass balance) was problematic because it involved several uncertain rate parameters such as denitrification and primary productivity.
- Method D (tidally averaged estuarine circulation model) had insufficient documentation and the panel recommended it be used as a consistency check only.
- Method A (Knudsen salt balance) was recommended because it relies on fewer estimated parameters. However, the panel recommended a re-analysis of layer salinity that incorporates the vertical profile of horizontal velocity and salinity.
- The Knudsen salt balance neglects tidal dispersion, which could be significant.
- Steady-state analyses neglect time-varying components that could induce significant exchanges.
- ORCA buoy data were not exploited to evaluate uncertainty in marine flux
- The dispersive flux of nitrogen to the euphotic zone was not included in aggregated model estimates (it was included in ROMS model estimates).
- Potential horizontal advective flux into the lower euphotic zone (below pycnocline) was missed in the nitrogen mass balance.
- The method used to identify the pycnocline depth (based on ORCA buoy data) was not documented.
- The method used to extrapolate near-surface salinity values from buoy data was not documented.

The independent panel also recommended use of a consistent 2-layer model construct for all calculations related to the marine nitrogen flux rather than the combination of 2- and 3-layer

Table 9. Available estimates for nitrogen loads to the surface layer and euphotic zone of Hood Canal and Lynch Cove

Study	Time Frame	Geo-graphic region	Marine Loading	Watershed Loading	Total Loading to Surface	% Watershed	Human Loading [8]	% Human	Shoreline OSS Loading	% Shoreline OSS
Hood Canal – Annual										
Paulson et al. (2006)	2004	Hood Canal	10,100 – 31,000 MT/yr	688 MT/yr	10,788 – 31,688 MT/yr	2% - 6%	NA	NA	26 MT/yr	< 1%
Steinberg et al. (2010)	2005-06	Hood Canal	39,000 MT/yr	750 MT/yr	39,750 MT/yr	2%	449 MT/yr	1%	NA[1]	<1%
Lynch Cove – Annual										
Steinberg et al. (2010)	2005-06	Lynch Cove	438 MT/yr	79 MT/yr	517 MT/yr	15%	73 MT/yr	14%	21 MT/yr	4%
Lynch Cove – Summer (loading to euphotic zone)										
Calculation from Twanoh buoy data	July – Sept 2006	Lynch Cove	168 MT/mo[4]	NA	NA	NA	NA	NA	NA	NA
Paulson et al. (2006)	Sept – Oct 2004	Lynch Cove	132 MT/mo[4]	3.5 MT/mo	135.5 MT/mo	3%	NA	NA	0.8 MT/mo	< 1%
Kawase and Bahng (2012)	July – Sept 2006	Lynch Cove	78 MT/mo[3]	NA[9]	NA	NA	NA	NA	NA	NA
Kawase and Bahng (2012)	Long term average	Lynch Cove	34 MT/mo[3]	NA[9]	NA	NA	NA	NA	NA	NA
Devol et al. (2011a)	July – Sept 2005-2006	Lynch Cove	11.1 – 57.3 MT/mo[5]	4.5 MT/mo[6]	15.6 - 61.8 MT/mo[6]	7% - 29%	3.6 MT/mo[6]	6% - 23%	2.6 MT/mo[6]	4% – 17%
Lynch Cove – Summer (loading to surface layer above pycnocline)										
Devol et al. (2011a)	July – Sept 2005-2006	Lynch Cove	5.3 – 19.2 MT/mo[2]	4.5 MT/mo[7]	9.8 - 23.6 MT/mo[7]	19% - 46%	3.6 MT/mo[7]	15% - 37%	2.6 MT/mo[7]	11% -27%

Notes:

[1]Steinberg includes only a hypothetical worst-case OSS calculation for entire watershed population indicating that OSS is <1% of marine loading

[2]Loading based on observed nitrogen concentrations at 6 m depth (interface between the top layer and middle layer in 3 layer construct). Advective flux only.

[3]Loading based on simulated conditions at 10 m depth. Sum of advective and diffusive fluxes. Advective flux was 37 MT/mo in 2006 (not reported for long term average simulation).

[4]Not explicitly characterized as loading to euphotic zone but as loading into Lynch Cove at depth (based on measured horizontal current velocities)

[5]Loading based on observed nitrogen concentrations at 12 m depth (interface between the middle layer and bottom layer in 3 layer construct). Advective flux only.

[6]Watershed and OSS loading assumed in Devol et al. (2011a) are drawn from Richey et al. (2010). As described in question 1, the OSS loads used in these calculations (2.6 MT/mo) exceed the range supported by seep measurements and groundwater flow estimates. Substituting a midrange value of 1.0 MT/mo from Table 8 produces the following values:
Total watershed loading = 2.9 MT/mo
Total loading (marine + watershed) = 14.0 – 60.2 MT/mo
Human loading = 2.0 MT/mo (3% - 14% of the total loading)
Shoreline OSS = 1.0 MT/mo (2% - 7% of the total loading)

[7]See footnote 6. Substituting a value of 1.0 MT/mo for shoreline OSS produces the following values:
Total watershed loading = 2.9 MT/mo
Total loading (marine + watershed) = 8.2 – 22.1 MT/mo
Human loading = 2.0 MT/mo (9% - 24% of the total loading)
Shoreline OSS = 1.0 MT/mo (5% - 12% of the total loading)

[8]All listed human loadings are calculated based on the assumption that all red alder contributions are human-caused. In reality, some fraction of these loadings are naturally occurring.

[9]The ROMS model application (Kawase and Bahng, 2012) included analysis of dissolved oxygen impact based on estimated nitrate concentrations in tributaries under natural and current conditions (80 µg/L and 150 µg/L, respectively). The nitrogen loadings for these simulations were not reported. See Question 3 below for further discussion.

models used by Devol et al. (2011a). Review panel comments indicated a preference for the euphotic zone to calculate the marine nitrogen flux.

EPA and Ecology concur with the panel's concerns with methodology and documentation, but follow-up discussion with Hood Canal researchers failed to yield a consensus on the path forward. The goal of this summary is to identify the "best estimates" for calculation of human impacts. It is clear from the independent review report that all of the available estimates for marine flux could be substantially improved through additional analysis and peer review. The panel recommended a full re-analysis of the marine nitrogen flux. EPA and Ecology would support re-analysis in the future, but this report is focused on existing research rather than planning new analyses. We have identified the currently available estimates in this report while providing appropriate caveats and concerns about methodology and documentation.

For analysis of dissolved oxygen impacts (see Question 3), we have conducted a Monte Carlo analysis of the aggregated model that includes uncertainty estimation. Reviewing the available estimates in light of the issues identified by the independent panel, we have identified a wide range of potential marine nitrogen fluxes as best available yet highly uncertain. The low end of the range is the advection-only flux to the surface mixed layer of Lynch Cove based on a standard salt balance (Method A) in Devol et al. (2011a). This value (5 MT/mo; see Table 9) is set as the low bound, because it does not include mixing due to tidal dispersion. For the high bound, we used the ROMS model estimate for 2006 in Kawase and Bahng (2011). The ROMS simulation includes dispersive fluxes, but it may over-estimate the total flux due to excess vertical mixing noted in the calibration (see further discussion of the model later in this document). The range based on these methods is 5 to 78 MT/mo. The high end of this range is lower than the estimated fluxes based on current meter observations in Paulson et al. (2006) and calculated herein based on the mean current velocity at the Twanoh buoy reported in Devol et al. (2011a). The large differences between these values for advection to the lower layer of Lynch Cove (132 – 168 MT/mo), ROMS model estimates (37 MT/mo), and the salt balance calculations (11 MT/mo) have not been adequately analyzed and resolved at this time.

As discussion of the available information continues with stakeholders, we will explore options for future work to improve marine nitrogen flux estimates that strongly influence the estimation of human impacts to dissolved oxygen in Lynch Cove.

Based on this review, the range of estimates for marine nitrogen loadings to the surface waters of Lynch Cove are provided in Table 10. Based on these loadings, we can calculate the relative contribution to surface layer from marine sources and watershed loading. The watershed loading can be further broken down into total human contributions and shoreline OSS contributions. Later, we use the range of marine loading estimates (5-78 MT/mo) in a Monte Carlo analysis of dissolved oxygen impacts (Question 3 below).

Table 10. Potential range of marine nitrogen loading to surface water in Lynch Cove and relative contribution of human loadings. Synthesis of available information for 2005-2006 summer conditions.

Human Impact Scenario	Marine Loading	Watershed Loading	Total Loading to Surface Layer	Watershed Human Loading	Fraction of Total Loading from Human Sources	Shoreline OSS Loading	Fraction of Total Loading from Shoreline OSS
	TDN[1] (MT/mo)	TDN (MT/mo)	TDN (MT/mo)	TDN (MT/mo)	%	TDN (MT/mo)	%
Lower Bound	78[2]	1.5[3]	79.5	1.3[4]	2%	0.3[5]	<1%
Upper Bound	5[1]	3.1[3]	8.1	2.9[4]	36%	1.9[5]	23%

[1] Value for mixed surface layer using Method A in Devol et al. (2011a).
[2] Value for 2006 euphotic zone loading from ROMS model in Kawase and Bahng (2012).
[3]Tributary estimates from statistical model in Steinberg et al. (2010) and repeated in Richey et al. (2010). Total tributary loading is 1.2 MT/mo (1.0 MT/mo from human sources). Lower and upper bound defined by shoreline OSS range of 0.3 to 1.9 MT/mo.
[4]The total red alder and human population contribution in watershed based on statistical model in Steinberg et al. (2010) and Richey et al. (2010) is 1.0 MT/mo, plus shoreline OSS loading of 0.3 to 1.9 MT/mo.
[5]From synthesis in Table 8.

The answers to Questions 1 (Watershed Nitrogen Loadings) and 2 (Marine Nitrogen Loadings) provide estimates of the relative contribution of human nitrogen loadings to the total nitrogen mass entering the surface layer of Hood Canal or Lynch Cove. Researchers used a variety of methods and estimated a range of magnitudes for both marine inputs and local human sources. All analyses suggest that the predominant overall source of nitrogen to Hood Canal is natural nitrogen entering from the Pacific Ocean at depth and entraining into the surface layer of Hood Canal through vertical mixing. The dominant contribution of marine nitrogen holds throughout the year and throughout central Hood Canal and Lynch Cove. Nevertheless, the state water quality standards require further analysis of the impact of human-caused loadings. Where natural conditions cause dissolved oxygen levels to fall below threshold concentrations (7.0 mg/L in Hood Canal), the total of all human sources cannot deplete oxygen levels by more than 0.2 mg/L for biologically relevant spatial and temporal scales. Estimates of oxygen depletion are assessed under Question 3 below.

Question 3: What is the impact of human nitrogen contributions on dissolved oxygen in Hood Canal?

This question is the most important and also the most difficult to answer, because a rigorous estimate requires reliable and comprehensive data on marine and terrestrial nutrient loading, oxygen concentration data over space and time, and a dynamic, 2- or 3-dimensional model of Hood Canal that is well tested against observations and shown to have good skill in simulating oxygen variability. Before delving into the modeling challenges, we summarize monitoring data that characterize long-term oxygen conditions in Hood Canal, including sediment core data and dissolved oxygen trend analyses. This is followed by a summary of impacts estimated using aggregated models and the 3-dimensional water quality model. Finally, we describe conditions leading to fish kill events and a potential connection between Lynch Cove and Hoodsport.

Background – Sediment Core Study

The NOAA Coastal Hypoxia Research Program funded a study by the Pacific Northwest National Laboratory and other organizations to collect and analyze sediment cores from Hood Canal. Sediment cores were dated using isotopic lead. Redox-sensitive metals were used to indicate historical oxygen conditions. In addition, the cores were analyzed for microfossils to identify phytoplankton assemblages. This work resulted in a report (Brandenberger et al., 2008) and journal article (Brandenberger et al., 2011). Key findings include:

1. Sediment cores indicate that hypoxia occurred in Hood Canal before European settlement.

2. Overall oxygen levels were lower in the 19th century than in the 20th century.

3. Anthropogenic forces throughout the 20th century have substantially increased the organic matter fluxes entering Hood Canal and Puget Sound. Trends in the late 20th century indicate a progression toward pre-industrial patterns in organic matter.

4. The cores show a strong shift from predominantly marine organic matter and lower oxygen conditions during the 18th to 19th centuries to more terrestrial organic matter with more oxygenated conditions during the 20th century.

In contrast to what was anticipated, Brandenberger et al. (2008) found that major land use changes in Puget Sound and Hood Canal watersheds did not coincide with decreasing marine oxygen conditions. On the contrary, low-oxygen conditions prevailed in a decadal pattern prior to the 1900s and reflected climate influences such as the Pacific Decadal Oscillation (PDO). Oxygen-rich conditions generally occurred mid-century.

Brandenberger et al. (2008) provides evidence of shifts in phytoplankton communities during urbanization. The microfossil reconstructions for diatom and foraminifera were too limited to support statistically valid conclusions. However, overall abundance of diatom

assemblages declined over time since circa 1900, with the most pronounced decline appearing after the 1950s (see Figure 39 of Brandenberger et al, 2008; HC-5 core). This trend is opposite to expected results related to eutrophication of the water column. However, the number of genera identified in the samples declined circa 1950s as planktonic species increased. These relationships may indicate a recent trending toward eutrophic conditions in the last several decades. Alternatively, the decrease in diatom abundance may confirm other paleoecological markers (organic markers, biogenic silica, and redox-sensitive metals) that indicate a decline in all proxies of water column productivity starting in the early 1900s and reaching a minimum in the 1950s.

While sediment core analysis is extremely valuable in providing information about long-term trends in water quality, the core samples cannot provide information about the most recent decades of hypoxia in Hood Canal. The cores also cannot determine whether additional human contributions have exacerbated the naturally low oxygen levels or whether they have led to low dissolved oxygen in shallow waters such as Lynch Cove.

Trends in Dissolved Oxygen Measurement Data

A long-term downward trend in dissolved oxygen over time would be one line of evidence indicating that population growth could be impacting Hood Canal dissolved oxygen concentrations. Dissolved oxygen in Hood Canal and elsewhere in Puget Sound and the Straits has been measured periodically since the 1950s.

Bassin et al. (2011) analyzed trends in the dissolved oxygen data for five locations in Puget Sound for the period from 1932 to 2009 using a seasonal Kendall test applied to decadal data. The five locations were the Strait of Juan de Fuca, Admiralty Inlet (traditional northern boundary to define Puget Sound), Point Jefferson in the main basin (Central Puget Sound), central Hood Canal, and Lynch Cove. The analysis did not include Ecology's ambient monitoring data collected in the 1970s and 1980s although data exist for four of the five locations. The analysis focused on dissolved oxygen concentrations at 100 to 200 meters depth from the first three stations, an oxygen inventory below 20 meters across six stations in the main arm of Hood Canal (Warner, 2011a), and concentrations at a depth of 20 meters in Lynch Cove. The Lynch Cove site is shallower than the other stations.

Bassin et al. (2011) found a statistically significant decreasing trend in dissolved oxygen concentrations from 1950 to 1999 in the Hood Canal (main arm) inventory (Figure 21). Because the Strait of Juan de Fuca and Admiralty Inlet also exhibited declining dissolved oxygen for the period 1950-2009, Bassin et al. (2011) did not rule out Pacific Ocean or other influences that are common to all stations. All are subject to human influences as well. The trends vary substantially depending on the time period used in the analyses. In fact, the Hood Canal oxygen inventory increased for the period 2000-2009, and Lynch Cove exhibited the fastest increasing rate for the same period. Temperature trends influence but do not account for the magnitude of dissolved oxygen change. There is no evidence of a unique, negative trend in the central Hood Canal oxygen concentrations compared with other sites in Puget Sound.

Many factors influence these trends, including the time period used for the analysis. As noted in Brandenberger et al. (2008), the 1950s data collection period coincided with a cool-phase Pacific Decadal Oscillation (PDO), which is associated with higher oxygen levels. More recent conditions include warm-phase PDOs associated with lower oxygen levels. Comparing data from 1930-66 with data over the last decade does not account for climate cycles. Because of this cyclic pattern, differences between the 1950s and present day cannot be solely attributed to humans causing a downward trend. In fact, the increasing trend in dissolved oxygen at Lynch Cove, where human contributions are expected to have the largest influence, run counter to population patterns.

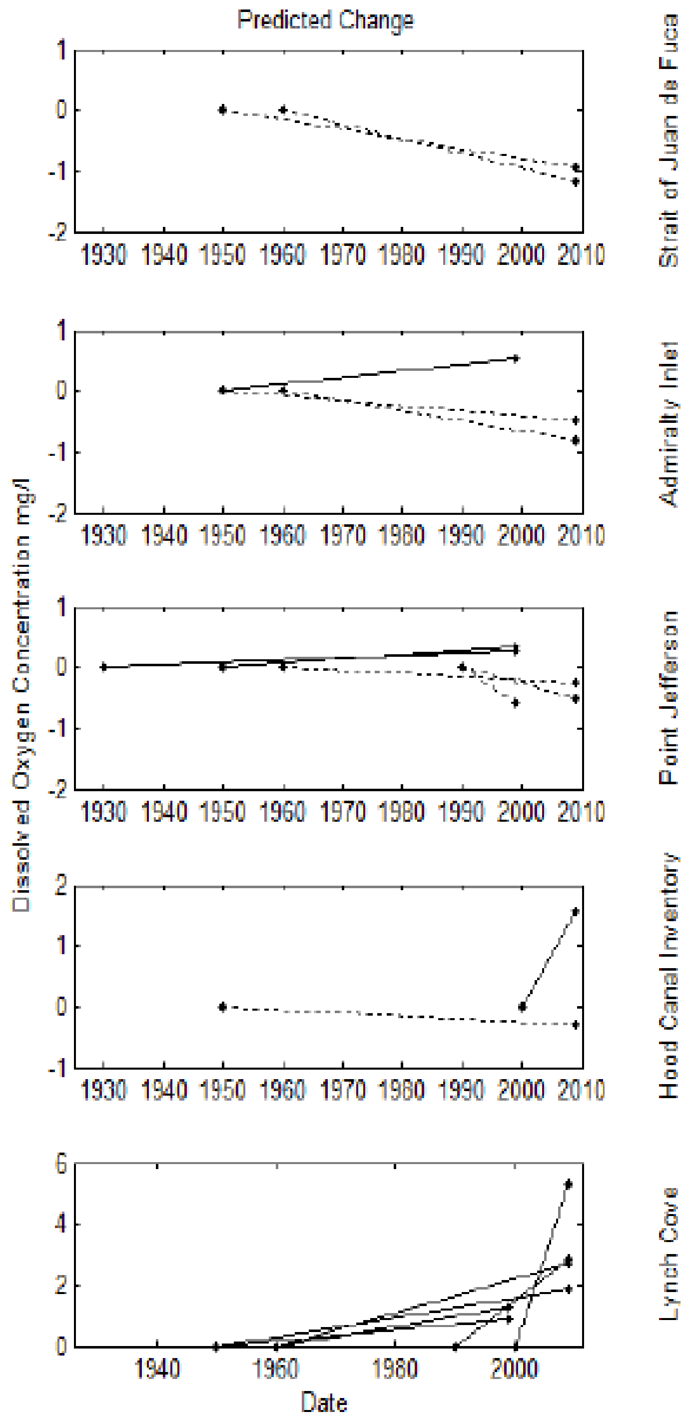
Based on review of the available information, we have found no compelling evidence that humans have caused decreasing trends in dissolved oxygen in Hood Canal. Climate cannot be ruled out, the trends are not unique to Hood Canal, and dissolved oxygen in Lynch Cove, where relative human influences are the largest, have increased and not decreased.

Oxygen Depletion Estimates based on Aggregated Models

Both aggregated model analyses (Devol et al. 2011a, Brett 2010a) and the ROMS biogeochemical model discussed in the next section were used to estimate dissolved oxygen impacts. The work to date has focused primarily on estimation of average summer impacts in Lynch Cove as a whole rather than maximum impacts at specific locations and times. There has also been a strong focus on OSS impacts and less emphasis on overall human impacts from the watershed as a whole.

Brett (2010a) and Devol et al. (2011a) used aggregated models to estimate the proportion of total nitrogen loadings entering Lynch Cove due to human activity, and then applied this proportion to the total observed dissolved oxygen deficit in Lynch Cove to estimate human impacts. The calculations do not account for all processes affecting dissolved oxygen and represent simplifications of complex systems. For example, circulation is represented as a simple two-layer system, whereas the current meter data shows a more complex pattern (e.g., Figure 5 in Devol et al. 2011b). Similarly, seasonal time frames are selected (e.g., June through September to represent peak population and algae growth) that may or may not account for all influences on the system (e.g., excluded spring nitrogen loadings may affect summer conditions and estimated impacts). Nevertheless, aggregated models can provide reasonable first approximations of human impact.

Figure 21. Statistically significant trends (seasonal Kendall) in dissolved oxygen at five locations in Puget Sound and the Strait of Juan de Fuca.



Source: Bassin et al. (2011)

Total Observed Dissolved Oxygen Deficit in Lynch Cove

Both Brett (2010a, 2011i) and Devol et al. (2011a) estimated the total dissolved oxygen deficit in Lynch Cove based on the data collected at the ORCA buoys at Twanoh and Hoodsport. Brett (2010a) found that average annual dissolved oxygen concentration below 11 meters depth was 1.1 mg/L less in Lynch Cove (Twanoh buoy) than central Hood Canal (Hoodsport and Duckabush buoys from 2005 to 2009 (see Figure 8 for a plot of this data). If Lynch Cove is compared to Hoodsport alone, the average difference was 0.7 mg/L (pers. comm. 2011i). In contrast, Devol et al. (2011a) estimated a total deficit of 1.6 to 2.3 mg/L based on the time series at the Twanoh and Hoodsport buoys.

Brett (pers. comm., 2011i) noted the significant disparity in the estimates of the oxygen deficit in Lynch Cove in Brett (2010a) and Devol et al. (2011a) despite the use of a common dataset from the Hood Canal ORCA buoys. Based on additional data comparisons, Brett (2011i) concluded that the biggest factor causing the disparity in deficit estimates was the depth range used in the data analysis. Central Hood Canal is significantly deeper (100 m at the Hoodsport buoy) than Lynch Cove (30 m at the Twanoh buoy). The choice of the depth range was based on differing assumptions about the connectivity of the deeper waters of Hoodsport (30 m to 100 m) to Lynch Cove (10 to 30 m). Brett (2010a, 2011a) used a depth range from 11 meters to the bottom, while Devol et al. (2011a) excluded values from 30 meters to the bottom at Hoodsport.

Devol (pers. comm., 2011d) analyzed the assumptions for mixing between Hoodsport and Lynch Cove. He noted that ocean water flows freely along isopycnal (equal-density) surfaces, and it takes much more energy to mix or flow across isopycnal zones. During the early and mid-summer, prior to the late-summer ocean intrusion, the density of water at 25 meters is approximately equal at Hoodsport and Twanoh (Lynch Cove). However, the density increases as depth increases at Hoodsport. It requires work (energy) to move this dense water up to a shallower depth. In the absence of an external force, Devol noted, it is unlikely that water at 20-30 meters depth in Lynch Cove either originates from or mixes with higher-density water at greater depths at Hoodsport. For this reason, he excluded oxygen values below 30 meters depth at Hoodsport.

The independent review panel recommended that the deficit calculation focus on oxygen conditions along similar densities (termed "isopycnal lines") between Hoodsport and Twanoh. Devol (pers. comm., 2011d) noted that this was the approach taken in Devol et al. (2011a). He elaborated that the isopycnal lines are horizontal between Hoodsport and Twanoh for most of the summer, and therefore common depths between the two locations should be used. This density pattern would call for exclusion of dissolved oxygen data from 30 meters to the bottom in Hoodsport from the comparison to Twanoh concentrations.

The aggregated modeling approach rests on an assumption that circulation conditions are constant over the period of analysis (June through September). For most of this period, it is reasonable to estimate circulation flows and mixing based on the assumption that mixing is constrained within isopycnal zones. However, the intrusion of denser, low oxygen marine water

at depth in September is a significant external forcing mechanism that alters the mid-summer balance. As Devol (2011d) notes, energy is required to overcome isopycnal mixing, and the September intrusion provides an influx of mixing energy. It is plausible that the early-summer dissolved oxygen deficit is best evaluated using the Devol et al. (2011a) approach, while the September deficit may be better characterized by the Brett (2011i) approach.

Devol et al. (2011a) filtered data using a 25-day lag to address the travel time between Hoodsport and Twanoh. It is not clear whether it is appropriate to incorporate a fixed time lag into the analysis, because the travel time likely varies significantly with the onset of the September intrusion. It is also difficult to discern a clear lag in time series comparisons (see Figure 8). Furthermore, it appears that the 25-day travel time calculated by Devol et al. (2011a) was based on circulation conditions during the September intrusion of saline ocean water, when current speeds are unusually high compared to mid-summer conditions. We back-calculated the velocity from the advective flow from Devol et al. (2011a) using the salt balance method. The resulting value (0.0005 m/s) is an order of magnitude lower than the velocity used in the travel time calculation by Devol et al. (2011a).

For the Monte Carlo analysis of aggregated model estimates described below, we adopted the oxygen deficit estimates of Devol et al. (2011a) that reflect stable isopycnal conditions in early summer (June through early August). Based on concerns about the appropriateness of the fixed lag assumption of 25 days, we selected the non-lagged values in Devol et al. (2011a). We used the 20-30 meter depth range, where the mean deficit was 1.6 mg/L with a standard deviation of 0.5 mg/L for 2006-2010. The deficit for the full depth range below the euphotic zone (12-30 meters) was not estimated in Devol et al. (2011a).

OSS Impacts on Dissolved Oxygen

Steinberg et al. (2010) used a two-layer salt balance to quantify the relative contribution of watershed and marine nitrogen loading on an annual basis (see Question 2 discussion). That analysis found that shoreline and tributary OSS contributed at most 0.5% of the total annual nitrogen loading to the main arm of Hood Canal and at most 8% of the annual loading to Lynch Cove based on the per capita estimates. Steinberg et al. (2010) did not estimate dissolved oxygen impacts associated with these human nitrogen contributions.

Subsequent analyses by Brett (2010a) used the estimates of the human contribution of nitrogen to the surface layer from OSS, along with dissolved oxygen data for Hood Canal, to estimate the human impact on dissolved oxygen from OSS. First, Brett (2010a) found that average annual dissolved oxygen concentration below 11 meters depth was 1.1 mg/L less in Lynch Cove (based on the Twanoh buoy) than central Hood Canal based on the Hoodsport and Duckabush buoys from 2005 to 2009. Comparing to Hoodsport alone, the difference is 0.7 mg/L (Brett, 2011i). Figure 8 presents this data. By multiplying a 4-8% nitrogen loading contribution (percentage of total annual nitrogen in surface layer attributable to septic systems from Steinberg et al. (2010) by a 1.1 mg/L dissolved oxygen change, Brett (2010a) estimated <0.07 mg/L impact from septic systems in Lynch Cove.

Devol et al. (2011a) also employed aggregated models to estimate dissolved oxygen impacts, but the analytical approach differed from that used by Brett (2010a). First, as noted under Question 2, Devol et al. (2011a) used a 3-layer model construct with different nitrogen loading assumptions than Brett (2010a). In addition, Devol et al. (2011a) used the higher Richey et al. (2010) estimates for OSS loading based on a higher population and wider buffer (Table 6). Because of these differences, the associated fraction of total nitrogen loading to the surface water from OSS was significantly higher in Devol et al. (2011a) than in Brett (2010a).

Devol et al. (2011a) also compared dissolved oxygen concentrations at Hoodsport and Lynch Cove, but they narrowed the time frame to the summer months (June through September) and restricted the comparison to water depths less than the maximum 30 meter depth of Lynch Cove rather than the entire depth at Hoodsport. Devol et al. (2011a) also incorporated a 25-day lag time in the analysis to account for travel time between Hoodsport and Twanoh. This analysis resulted in estimated summer dissolved oxygen deficits in Lynch Cove, relative to Hoodsport, ranging from 1.2 to 2.6 mg/L for water depths of 6 to 30 m.

Devol et al. (2011a) calculated the human dissolved oxygen impact in Lynch Cove using an estimated total dissolved oxygen deficit of 2 mg/L. By multiplying this value by their estimate of the OSS fraction of total nitrogen entering the surface layer of Lynch Cove in summer (12 to 30% based on OSS load of 2.9 MT/mo from Richey et al. (2011); see Question 2), Devol et al. (2011a) estimated that the impact of OSS loading to dissolved oxygen concentrations was 0.24 to 0.60 mg/L.

In a separate analysis, Devol et al. (2011a; see also Newton and Devol, pers. comm., 2011) calculated the potential dissolved oxygen impact from OSS loading to Lynch Cove using stoichiometric ratios in phytoplankton. The authors considered this a check rather than a prediction of impact (Newton and Devol, pers. comm., 2011. Assuming all of the OSS loading produces phytoplankton biomass that sinks to the lower water column, Newton and Devol (pers. comm., 2011) calculate that the respiration of 2.9 MT/mo attributed to OSS nitrogen (from Richey et al., 2011) will result in an oxygen impact of 0.3 mg/L.

Table 11 lists the available estimates of OSS impacts on Lynch Cove dissolved oxygen concentrations in the summer from available studies. As noted under Question 2, the most representative groundwater measurements, water balance, and per capita estimates for Lynch Cove indicate a significantly lower shoreline OSS loading than the values reported in several studies to date. In the next section, we analyze the effect of these lower OSS loading estimates on dissolved oxygen deficits and the uncertainty in the prediction.

Table 11: Estimates of OSS impacts on dissolved oxygen in Lynch Cove in summer (all values are means for June through September)

Study	Method	Proportion of Surface Nitrogen Loading from OSS Sources (%)	Lynch Cove Total Dissolved Oxygen Deficit (mg/L)	OSS Impact to Dissolved Oxygen Deficit[1] (mg/L)
Brett (2010a) and Steinberg et al.(2010)	Proportion of Observed O_2 Deficit	4% - 8%[2]	1.1	0.07[2]
Devol et al. (2011a)	Proportion of Observed O_2 Deficit	12% - 30%[3]	2.0	0.2 - 0.6[3]
Devol et al. (2011a) and Newton and Devol (pers. comm., 2011)	Stoichiometric Analysis of Biomass Production/Decay	NA	NA	0.3[3]

[1]Combined shoreline and upland OSS contributions. These studies focused solely on impacts of OSS loadings rather than total human impacts. Total impacts would include population-related loadings and the human-caused portion of the red alder loadings in tributaries.

[2]These values calculated using an OSS loading of 1.8 MT/mo (see table 6).

[3]These values were calculated using an OSS loading of 2.9 MT/mo (2.6 MT/mo from shoreline OSS and 0.3 MT/mo from other watershed OSS; see Richey et al. (2011)).

Probabilistic Estimation using Aggregated Model

None of the studies published to date have attempted to compile a probabilistic assessment of the impact of human nutrient loading on marine dissolved oxygen. The preceding sections describe the "best estimates" of loadings and oxygen deficits in Lynch Cove, and generally describe the uncertainty in those estimates. To supplement the available work, we have estimated dissolved oxygen impacts using the aggregated model coupled with a Monte Carlo approach. The following equation represents the aggregated model calculation for human impact to dissolved oxygen:

$$Y = (N_{human}/N_{total}) * D$$

where,

Y = Human impact on DO (mg/L)
N_{human} = Human Loading of TDN to Lynch Cove surface layer (MT/mo)
N_{total} = Total Loading of TDN to Lynch Cove surface layer (MT/mo)
D = Dissolved oxygen deficit in Lynch Cove (mg/L)

This can be expanded to identify the individual TDN loading components estimated by researchers to date:

$$Y = ((N_{tribH} + N_{OSS})/(N_{tribH} + N_{tribN} + N_{OSS} + N_{marine})) * D$$

where,

N_{tribH} = Human Loading of TDN in tributaries (MT/mo)
N_{tribN} = Natural Loading of TDN in tributaries (MT/mo)
N_{OSS} = OSS Loading (MT/mo)
N_{marine} = Marine Loading (MT/mo)

Rather than using the single value estimates alone, the Monte Carlo approach employs a parameter distribution for each of the terms on the right side of this equation. Different sources and types of uncertainty are represented by the selected distributions, but these distributions do not account for all of the sources and types of uncertainty. Nevertheless, this approach provides an initial estimate of the potential range of uncertainty. In general, when considering multiple sources of uncertainty, we adopted the largest range among the potential sources of uncertainty. For example, for nutrient loadings from tributaries, we considered potential analytical error in measurements of nutrients and tributary flow as well as the correlation error of the statistical model used to estimate the origin of the nutrients (e.g., humans v. natural sources). In this case, we selected a typical range of uncertainty in laboratory analyses for nutrients (+/- 20%), because this uncertainty was higher than the uncertainty in flow and model correlation. which is also influenced by laboratory analysis error. The distribution characteristics are listed in Table 12. Selection of the characteristics of the parameter distributions was based on best professional judgment as described in the notes.

Table 12: Assumed distributions for the uncertainty analysis

Para-meter	Distribution	Units	Mean	Std Dev	Range	Uncertainty Represented
N_{tribH}	Normal	MT/mo	1.0	0.20	NA	General estimate of analytical error for nutrien concentrations
N_{tribN}	Normal	MT/mo	0.2	0.04	NA	General estimate of analytical error for nutrien concentrations
N_{OSS} (Flow term)	Uniform	m^3/sec	NA	NA	0.5 – 1.1	Range of groundwater flow estimates (Brett (2 Paulson et al. (2006))
N_{OSS} (TDN term)	Random Sampling of Data	mg/L	NA	NA	NA	Variability of DIN concentration in 325 samp shoreline seeps in Hood Canal (Sheibley and I pers. comm.)
N_{marine}	Uniform	MT/mo	NA	NA	18 - 78	Range of estimates based on available models (2011a), Kawase and Bahng (2012))
D	Normal	mg/L	1.6	0.5	NA	Mean and variance of deficit estimates in Dev (2011a)

Notes:

(1) For the tributary estimates, the mean value is taken from summer loading estimate from the statistical model of Steinberg et al. (2010). We selected a standard deviation value of 20% around the mean. This is a rough approximation of the error in laboratory analyses for nutrients. This choice is based on the assumption that uncertainty/error in nutrient concentration is the primary source of uncertainty in the nutrient loading estimation, rather than the uncertainty in flow estimates.

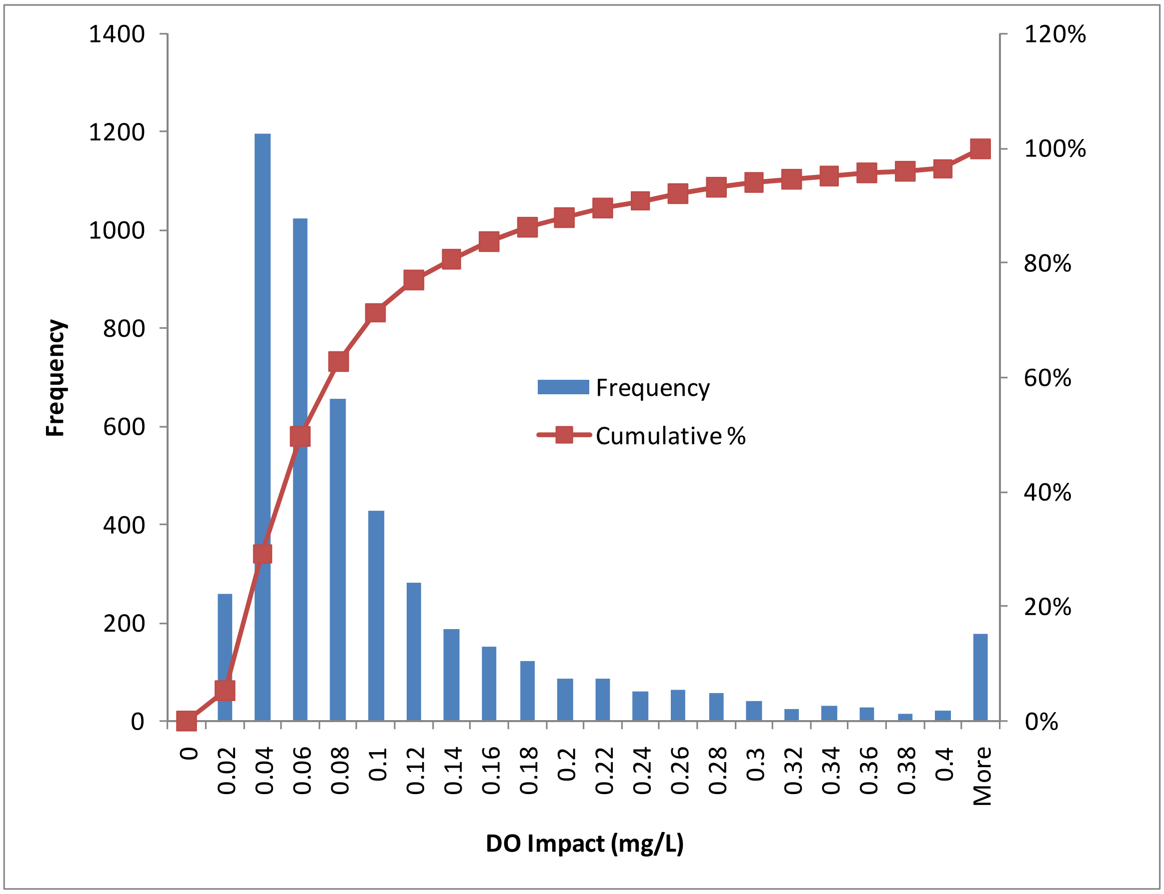
(2) For the OSS loading, the flow distribution is uniform, with the range based on estimates of Brett (2011d) and Paulson et al. (2006). For TDN concentration, there are sufficient data in the Hood Canal shoreline sampling data (325 seep samples) to randomly sample from the data instead of using a parameterized distribution. The Hood Canal data was selected over the data from Lynch Cove sites based the scarcity of Lynch Cove data (23 samples) and the similarity in the median values of the two datasets.

(3) For the marine flux term, we have noted the significant differences among researchers in assumptions and methods as well as concerns about methodology identified by the independent review panel. For this analysis, we simply identified the full range of available estimates and assumed a uniform distribution, which assumes an equal probability of occurrence of any value in the range.

(4) For the Lynch Cove oxygen deficit term, we have used the average and standard deviation of multiple-year estimates in Devol et al. (2011a) derived from spatial and temporal patterns in marine dissolved oxygen concentrations.

The Monte Carlo simulation was run with 5000 random samples drawn from the distributions and groundwater seep data. The resulting distribution for the human impact estimate is shown in Figure 22. The median predicted impact is 0.06 mg/L and the 10^{th} and 90^{th} rank percentile predictions are 0.03 mg/L and 0.23 mg/L, respectively.

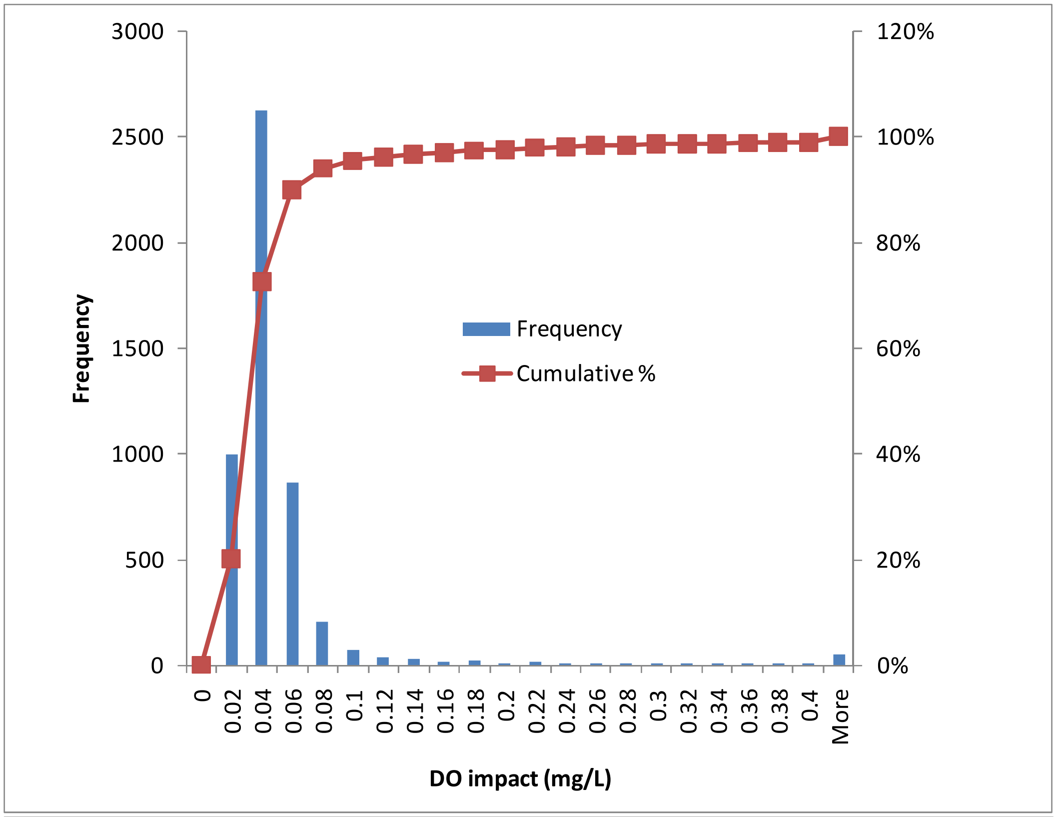
Figure 22: Lynch Cove Impacts based on Monte Carlo Analysis.

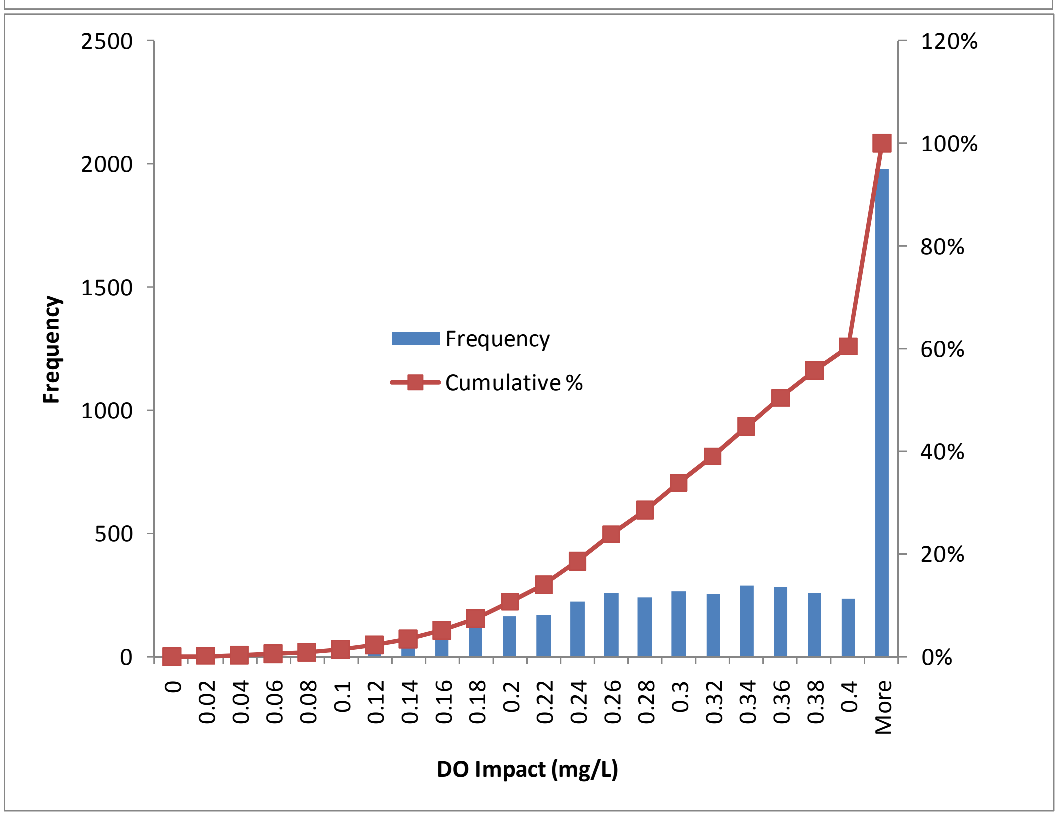


Sensitivity of Monte Carlo Results to Marine Flux Range

The results above are strongly dependent on the estimated range of uncertainty in the flux of marine nitrogen into Lynch Cove (5 – 78 MT/mo). This sensitivity is evident when we apply the Monte Carlo simulation with fixed values for the marine flux at the low and high bound. A side-by-side comparison of these two sensitivity tests is shown in Figure 23.

Figure 23: Sensitivity of Impact Estimates to Marine Flux. Results with Marine Flux at High Bound (78 MT/mo; top plot) and Low Bound (5 MT/mo; bottom plot)





In summary, each of the aggregated model calculations relies on key assumptions and data derivations that impose uncertainty in the resulting estimates. The aggregated model method, where the relative human nitrogen contribution to total nitrogen loadings is combined with observed oxygen deficits between central Hood Canal and Lynch Cove, is a reasonable screening-level approach for coarse spatial and temporal scales. Since the method relies on a simplification of circulation patterns and seasonal processes, the resulting estimates should be considered first approximations. All of these approaches employ an averaging of system characteristics over long time spans (seasonal/annual) and space (Lynch Cove/Hood Canal and different portions of the water column), so they provide estimates of mean impact. Based on a synthesis of available estimates and the median estimate from a Monte Carlo analysis, we estimate that the mean human impact in the summer in Lynch Cove is approximately 0.06 mg/L (range of 0.3 to 0.23 mg/L). However, due to the large uncertainty in marine nitrogen loading and other limitations in the available information, this is highly uncertain. The results are inconclusive with respect to compliance with water quality standards.

Estimates based on ROMS Model Predictions

Because of the complex processes that affect dissolved oxygen conditions in marine waterbodies, researchers and regulators often use water quality models to estimate human impacts to ambient conditions. The most sophisticated models provide continuous simulations of the natural processes that affect nutrients, algae, and dissolved oxygen in a water body. The use of spatially explicit numerical models is very common in supporting management actions under the Clean Water Act. Puget Sound, by virtue of its complex bathymetry, presents a unique challenge for the model developer. Unlike a river, which can be effectively analyzed using a 1-dimensional, steady-state model, Puget Sound analysis requires development and application of a 2- or 3-dimensional, time-varying model.

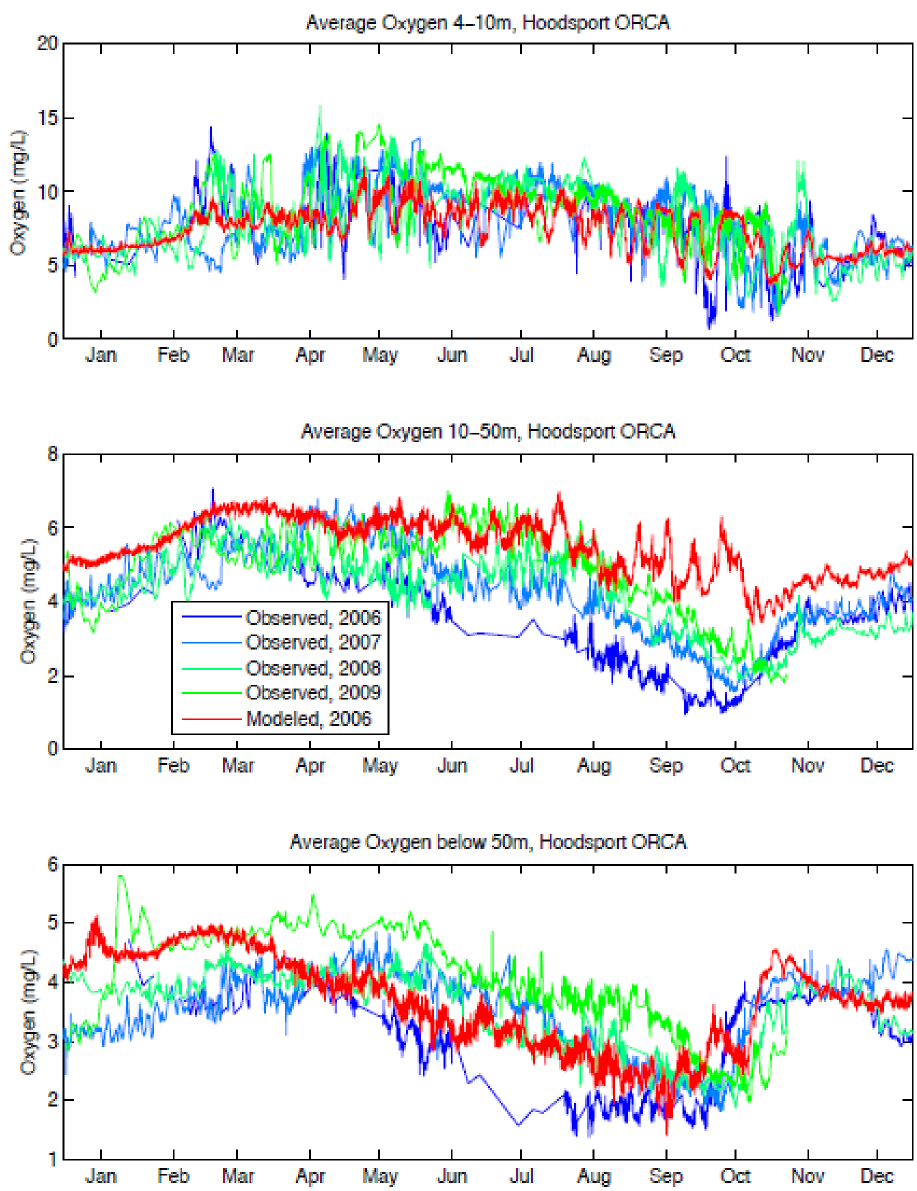
The ROMS model is the most comprehensive analytical tool used to date to estimate dissolved oxygen impacts in Hood Canal, because it can incorporate all major processes that affect oxygen, including boundary conditions, seasonal variability, vertical and horizontal mixing, phytoplankton dynamics, and nitrogen loading, fate and transport. A water quality model essentially consists of two models in one, because it must simulate both water movement (circulation) and water quality. The two sub-models are typically developed in sequence starting with the circulation model.

Kawase (2007) and Kawase and Bahng (2010) documented the performance and remaining tasks for the ROMS hydrodynamic model. Kawase and Bahng (2012) describe the water quality model of Hood Canal and a set of initial model scenarios. This report is a reasonable first step in documenting model performance and application to initial model scenarios. However, insufficient information is provided to complete a full review of important model components for regulatory decision-making.

While the current documentation is limited, the report does include a number of plots comparing observed and simulated water quality constituents, computed error statistics, and discussion of model capabilities and limitations. In general, this information indicates that the model simulates the major processes affecting dissolved oxygen and provides plausible

predictions of average ("climatological") water quality conditions for constituents that affect dissolved oxygen. At the same time, the report highlights areas where the low model skill is cause for concern. For example, higher-than-measured salinity in the surface layer and oxygen at mid-range depths (see Figure 24 below) could indicate problems with circulation and mixing. Kawase and Bahng (2012) state that these errors would generally lead to an underprediction of dissolved oxygen impacts. The root mean square error for dissolved oxygen predictions ranged from 0.4 mg/L (Hoodsport at depth) to 1.7 mg/L (Lynch Cove surface layer). In general, model error decreased with depth.

Figure 24: Comparison of predicted and measured dissolved oxygen conditions at Hoodsport



Source: Kawase and Bahng, 2012

Kawase and Bahng (2012) ran three simple model scenarios using the climatological boundary conditions to evaluate the sensitivity of Hood Canal to changes in the nitrogen concentration of the tributaries. These simulations do not explicitly include OSS loadings as boundary inputs. All tributaries throughout central Hood Canal and Lynch Cove were set to a uniform, fixed concentration for nitrate-nitrite. The three selected concentrations were 80, 500, and 1,000 µg/L. The lowest value (80 µg/L) represented a "pristine" tributary, while 500 µg/L represented "impacted" tributaries. Since most tributaries are well below 500 µg/L, including large tributaries like the Skokomish River, this second scenario is arguably well beyond "worst case" for the Canal as a whole. The third scenario sets all tributaries to 1,000 µg/L ("severely impacted"). It is highly unlikely that all tributaries throughout Hood Canal could reach this concentration in the foreseeable future.

Kawase and Bahng (2012) analyzed the results of the three tributary scenarios and determined that the change in the predicted oxygen concentration was linearly proportional to the tributary nitrate concentration. This is consistent with the assumptions used in the proportional calculations and aggregated models (Devol et al., 2011a; Steinberg et al., 2010; and Brett 2010a). This relationship enabled Kawase and Bahng (2012) to estimate impacts from any nitrate concentration between 80 and 1,000 µg/L without re-running the model. In order to provide a more realistic estimate of current impacts to Hood Canal, they applied linear interpolation to the model output data to estimate the oxygen conditions that would occur with tributaries discharging at 150 µg/L, based on the annual, flow-weighted average nitrate concentration for tributaries to Hood Canal (Brett 2011b). After identifying the model grid cell with the highest impact, which was located on the southeast corner of Lynch Cove, they calculated a 10-day average impact at this location (Kawase, pers. comm., 2011g). Using this methodology, Kawase and Bahng (2012) estimated that the maximum dissolved oxygen impact was 0.14 mg/L in Lynch Cove.

Kawase (pers. comm., 2011a) provided additional modeling information for Lynch Cove to clarify findings presented in Kawase and Bahng (2012). This focused on model outputs for the difference in dissolved oxygen between the "pristine" scenario (tributaries set to 80 µg/L nitrate-nitrite) and the "impacted" scenarios (tributaries at 150 µg/L and 500 µg/L nitrate-nitrite). To facilitate comparison with the aggregated model results described above, Kawase (pers. comm., 2011a) averaged the differences for all Lynch Cove model cells for depths 10 meters to the bottom for each month. Table 13 presents these model outputs.

Table 13. ROMS model predictions for average dissolved oxygen impact in Lynch Cove at depths greater than 10 meters. Values refer to differences in dissolved oxygen compared with the 80 µg/L run and are compiled to compare with aggregated model results.

Tributary Loading Scenario	June through September mean dissolved oxygen impact (mg/L)
150 µg/L	0.04
500 µg/L	0.3

The 150 µg/L scenario likely underestimates the actual impact in Lynch Cove for two reasons. First, the model simulations did not explicitly include OSS discharges, which contribute to human nitrogen loading in the summer in Lynch Cove. Second, Lynch Cove tributaries average about 300 µg/L in the summer months (Figure 15), higher than the 150 µg/L scenario.

The 500 µg/L scenario likely overestimates the actual impact to Lynch Cove by setting the tributaries to a significantly higher concentration than current conditions (Figure 15). The scenario also sets the central Hood Canal tributaries, including the nearby Skokomish River, to 500 µg/L. This is far higher than current concentrations (about 100 µg/L, Figure 15), and would overestimate dissolved oxygen depletion at depth in central Hood Canal.

Considering current tributary concentrations and OSS loads, the 150 and 500 µg/L scenarios likely bracket the current watershed contributions. Based on available information, the 80 µg/L scenario represents a reasonable natural background condition. Because dissolved oxygen impact increases linearly with differential nitrogen load, the model scenario results indicate that the average summer impact on Lynch Cove dissolved oxygen lies within the range of 0.04 to 0.30 mg/L.

The ROMS model was not developed to a level of refinement sufficient to resolve the date, location, and magnitude of the maximum human impact. The model domain did not include portions of Lynch Cove with a depth less than 10 meters, and this truncated a substantial area of the Cove (Kawase and Bahng, 2010). Lynch Cove extends further east of the model grid boundary and becomes gradually shallower. This tapering will change the pattern of oxygen concentrations and impacts. Nevertheless, the model did provide general insights about mean and maximum impacts. The model predicted that June was the month with the highest monthly average impact. The predicted impacts in June were approximately 50% higher than the average impact for June through September (Kawase, pers. comm., 2011a). In addition, model predictions indicated that the maximum impact occurred at the eastern end of Lynch Cove, which was the area with the model grid limitations noted above (Kawase, pers. comm., 2011b).

If maximum impacts are assessed in the future, the modeling work should focus on constructing a more extensive and refined model representation of this area of Lynch Cove.

Synthesis of Human Impact Estimates

The available estimates of human impact to dissolved oxygen in Hood Canal and Lynch Cove have been produced using both aggregated box model calculations and the ROMS model. If all methods are reasonably implemented, these methods should be in general agreement on overall scale of impact (e.g., long-term average impact over large areas).

Table 14 provides a comparison of "best estimates" from aggregated model results and ROMS model predictions for the human impact to dissolved oxygen in Lynch Cove. While the aggregated and ROMS model analyses were not fully coordinated in terms of watershed loads, spatial areas, or time periods, the results are in general agreement.

Table 14. Comparison of predicted human impact on average summer[1] dissolved oxygen concentration in Lynch Cove below the euphotic zone[2].

Analytical Tool	**Estimated Human Impact (mg/L DO)**	
	Median	**Range**
Aggregated Model	0.06	0.03 - 0.23
ROMS Model	NA	0.04 - 0.3

[1] June through September.

[2] Depth of euphotic zone varies among researchers.

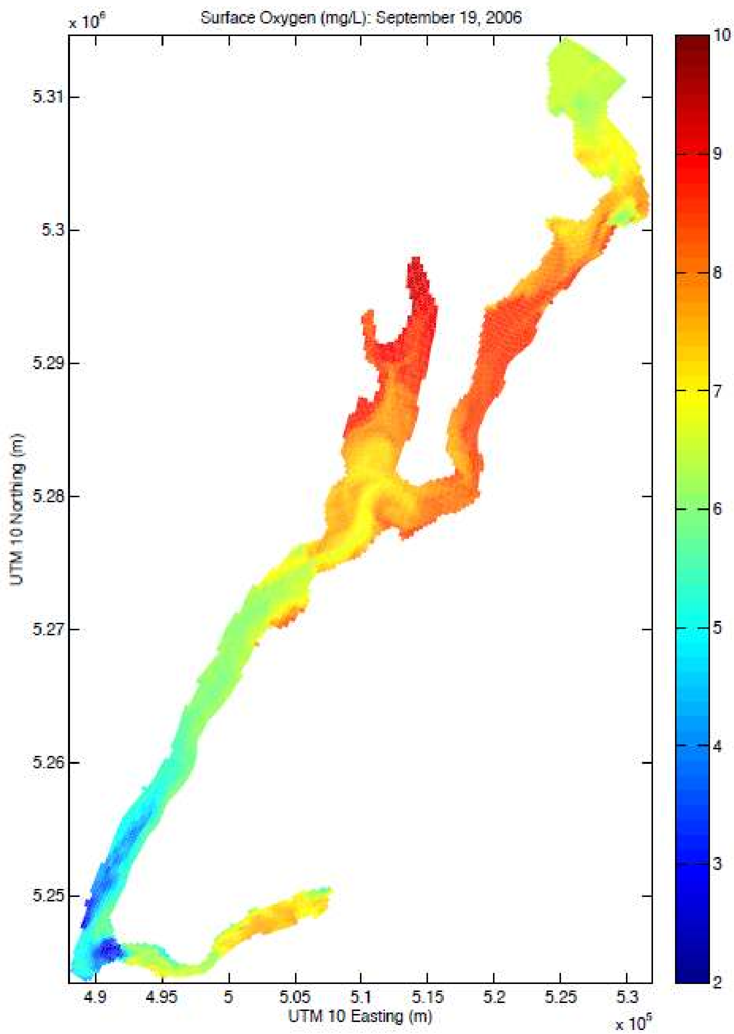
In summary, ROMS model simulations and aggregated models coalesce around an average summer impact ranging from 0.03 to 0.3 mg/L. As noted earlier, the Washington water quality standards restrict human impacts to 0.2 mg/L or less. The available models and analyses have limitations that preclude a reliable estimate of impacts. Therefore, there is not sufficient information to conclusively evaluate compliance with water quality standards at this time.

Analysis of Fish Kill Events

As noted earlier, the simple calculation methods were only employed in Lynch Cove, but the ROMS model was applied in central Hood Canal as well. The aggregated models were not used to assess the human contribution to low dissolved oxygen where fish kills occur. Kawase

and Bahng (2012) provide impact estimates for the area in central Hood Canal between Hamma Hamma and Annas Bay. This would correspond to the zone for fish kills. The maximum impact from human sources was 0.04 mg/L in this area, which occurred in June, whereas fish kills have occurred in September. Kawase and Bahng (2012) concluded that outside of Lynch Cove, the "impact of nutrient loading from impacted streams is still minuscule and is not a significant factor in issues such as fish kills in comparison with natural processes." Other analyses of fish kills in September 2006 and September 2010 provide strong evidence that southwesterly wind events cause sudden surfacing of low oxygen water to the surface layer in central Hood Canal, and that these wind events are the proximate cause of the fish kills (Kawase, 2007; Devol et al., 2011; Newton, 2010; Newton et al. 2011a; Kawase and Bahng, 2012). The hypothesis is supported by model simulations for September 2006 that predicted reduced dissolved oxygen at the surface of central Hood Canal at the fish kill location under a southwesterly wind (see Figure 25).

Figure 25: Water quality model prediction of surface oxygen during 2006 fish kill



Source: Kawase and Bahng, 2012

Connectivity between Hoodsport and Lynch Cove

Lynch Cove is the part of Hood Canal where human contributions have the greatest influence on dissolved oxygen levels, and the most recent analyses of human impacts have focused on this region. Three analyses (Paulson et al., 2006; Steinberg et al., 2010; and Kawase and Bahng, 2012) found no substantive link between human nutrient contributions and fish kills near Hoodsport. However, Mickett et al. (2011) documented a subsurface seaward outflow that potentially connects the region where humans have the greatest influence on dissolved oxygen (Lynch Cove) to the area and season of the fish kills (Hoodsport).

Three processes potentially influence water column oxygen levels at Hoodsport: (1) near-bottom dissolved oxygen declines over the summer and fall due to sediment and lower water column processes, (2) mid-depth dissolved oxygen declines due to water column respiration, and (3) low dissolved oxygen water at mid-depth is transported from areas landward by the subsurface seaward outflow from August through October.

First, ORCA buoy data do confirm that near-bottom processes draw down dissolved oxygen throughout Hood Canal over the summer. However, Mickett et al. (2011) ruled out local near-bottom processes since the near-bottom waters were not as low in dissolved oxygen as the minimum values at mid-depths at Hoodsport. However, this does not rule out near-bottom water from other regions transported seaward at mid-depths.

Second, water-column respiration depletes oxygen levels immediately below the euphotic zone. As described in Devol et al. (2011a), primary productivity in the euphotic zone produces organic matter that decomposes lower in the water column. Mickett et al. (2011, and personal communication) ruled out respiration as causing the dissolved oxygen minimum at 20 meters depth based on bounding calculations that have not been published. However, dissolved oxygen time series indicate that concentrations decline by 2 mg/L over 4 months (Devol et al., 2011a). Until additional respiration-based calculations are made available, water-column respiration, particularly during the late summer months, cannot be ruled out as a controlling factor for the dissolved oxygen minima observed at Hoodsport.

The subsurface seaward outflow represents the third potential factor influencing the dissolved oxygen minimum at Hoodsport. Three aspects of this phenomenon are described below: (1) evidence from current velocity data, (2) source of water transported, and (3) volume of water transported seaward at Hoodsport.

Current velocity data indicate a subsurface seaward outflow at Hoodsport, corroborated by patterns found at the Twanoh buoy and thalweg transects between the buoys conducted in 2006 and 2007. This subsurface seaward outflow appears seasonally, varies from year to year, and intensifies at weekly time scales. Mickett et al. (2011) links this subsurface seaward outflow to intrusions of high-density ocean water entering Hood Canal over the sill near Bangor. Density varies with coastal upwelling conditions, which affects how the intrusion propagates landward. Some proportion of the intrusion may propagate into Lynch Cove where it entrains water, rises to a specific density layer, then moves seaward via a subsurface outflow (July 26, 2011 meeting with Cope, Roberts, Mickett, Devol, Warner, Brett, and Newton). The timing coincides with

low-oxygen levels at Hoodsport but also occurs at other times of the year when fish kills and low dissolved oxygen do not occur. Kawase (pers. comm., 2011f) reports a persistent outflow at depth around Hoodsport simulated by the ROMS model.

The source of this water cannot be identified definitively based on the information available but is inconsistent with Lynch Cove density at the time that minimum dissolved oxygen levels occur. Current velocity and density data indicate that the water source is landward of Hoodsport and from depths shallower than 50 m (Mickett et al., 2011) but cannot isolate the region more specifically. The densities associated with the initial dissolved oxygen minimum at Potlatch in August 2006 propagated there from the landward direction and could have originated in Lynch Cove itself (Mickett et al., 2011 and Warner, pers. comm., 2011b). However, by September 2006, the density was greater than that measured at Sisters (Mickett et al., 2011; Warner, pers. comm., 2011b). Therefore, the water source feeding the subsurface seaward outflow identified at Hoodsport changed as dissolved oxygen continued to decline and was consistent with a shift to upwelled deep water (Warner, pers. comm., 2011c). In 2007, the minimum dissolved oxygen at Potlatch also indicates a denser, more seaward source than the water at Sisters Point.

Estimates of the volume of water transported by the subsurface seaward outflow vary widely among researchers. These are expressed as the ratio of low dissolved oxygen source water at Hoodsport to water transported by the subsurface seaward outflow. Mickett et al. (2011) estimates a ratio of 3 or 4 to 1, meaning the volume of water transported from landward of Hoodsport is 1/3 to 1/4 the volume of low dissolved oxygen water at Hoodsport. Details of the calculation are not presented. Brett (pers. comm., 2011j) estimates the ratio of Lynch Cove water volume to the volume of near-bottom low-dissolved oxygen water seaward of Sisters into the central Hood Canal as 1:25. However, this estimate includes more than just the water volume associated with the subsurface seaward outflow. Finally, Warner (pers. comm., July 26, 2011 meeting) estimates the ratio of entrained low dissolved oxygen water from landward of Sisters to the low dissolved oxygen water identified at Hoodsport as 1:7; the details have not been published. All three researchers use different approaches to estimate the ratios. None of the estimates suggest that the low dissolved oxygen source water transported to Hoodsport is more than 33% of the total volume transported by the subsurface seaward outflow. Brett (pers. comm., 2011i) also noted that the subsurface seaward outflow mainly occurred mid-thalweg and not in nearshore areas where fish kills have been observed.

In summary, the current velocity data indicate that an episodic subsurface seaward outflow occurs from August through October. Some instances of the outflow coincide in time and space with lowest dissolved oxygen conditions. The source of the low dissolved oxygen water cannot be identified definitively. However, several analyses contradict a link between the subsurface seaward outflow and fish kills:

- The subsurface seaward outflow affects the thalweg, or center, of the channel, and not the nearshore regions where fish kills occur.
- The ROMS model found a maximum 0.04 mg/L human impact on dissolved oxygen where fish kills occur, well below the level that would make any difference to the biota.

- The volume in Lynch Cove is insufficient to account for the volume of low dissolved oxygen water at Hoodsport.
- The density patterns suggest that the source of the water in the subsurface seaward outflow changes between August and September due to the inflowing marine waters.

While Lynch Cove may have contributed to the Hoodsport dissolved oxygen minimum in August, by September density signatures were consistent with upwelled near-bottom water and not Lynch Cove water. This indicates that Lynch Cove water is not influencing dissolved oxygen conditions at Hoodsport in the time frame (September) of fish kill events. The independent review panel concluded that "the evidence that low DO water from LC (Lynch Cove) makes a contribution to fish kills at Hoodsport is weak. Therefore, available information indicates that human nitrogen releases to Lynch Cove do not significantly contribute to fish kills at Hoodsport.

Uncertainty

Overview

The complex linkage between nutrient loadings and phytoplankton growth is a fundamental source of uncertainty in the analysis of nutrient effects on hypoxia. While researchers agree on the basic linkage of nitrogen loading, phytoplankton production, and dissolved oxygen depletion, phytoplankton productivity is highly complex and variable. The independent review panel highlighted this fundamental uncertainty in its report:

> *Phytoplankton community enumeration data from the Hood Canal (Pacific Shellfish Institute, 2008, 2009 progress reports) indicate that the phytoplankton biomass responses to environmental factors (including N inputs) are dominated by episodic blooms. These blooms do not always quantitatively track N inputs, but rather appear to be a response to several environmental factors that contemporaneously control their formation. These physical controlling factors include light, temperature, flushing and residence times, and vertical mixing, in addition to grazing. Therefore, even though N is likely to be the main nutrient that controls (limits) phytoplankton production, it is difficult to develop a strong and predictable direct relationship between N inputs and phytoplankton growth/bloom responses, especially over the time frame (days to weeks) that blooms typically develop, die and sink into bottom waters where they could fuel hypoxia.*

The effort to quantify the impact of anthropogenic nitrogen loadings on dissolved oxygen must be conducted with an acknowledgement of the complexity and variability of the aquatic environment.

Marine Models

All of the estimates of human impact are derived from quantitative water quality models of Hood Canal or Lynch Cove. These models range from steady-state aggregated models (e.g., 3-layer model of Devol et al., 2011a) to a high-resolution, 3-dimensional water quality model that simulates nitrogen, associated phytoplankton biomass, and dissolved oxygen concentrations (Kawase and Bahng, 2012). Both simple and complex models play an important role in diagnosing and analyzing water quality impacts.

Aggregated models

As noted earlier, aggregated box models are highly simplified representations of water quality processes, circulation patterns, and seasonal variation. The various aggregated models developed for Hood Canal are screening-level tools appropriate as first approximations. Aggregated models employ an averaging of system characteristics over long time spans (seasonal/annual), large areas (Lynch Cove/Hood Canal), and portions of the water column (surface mixed layer). By definition, they will only provide estimates of mean impact over large areas and seasonal time frames.

This report and the independent review (PSI, 2012) identified several fundamental limitations of the aggregated models applied to date. Because of the concerns with the methods, problems with the data used to derive the estimates of human impacts, uncertainty and the large range of marine nitrogen loading, and averaging over coarse spatial and temporal scales that may miss critical periods or areas relevant to aquatic life, we do not recommend that the aggregated models be compared with the water quality standards for regulatory purposes. Uncertainty renders the Lynch Cove aggregated model results inconclusive.

ROMS water quality model

Prediction error occurs in all water quality models, but it is particularly difficult to calibrate the nutrient, phytoplankton, zooplankton, and dissolved oxygen interactions in a dynamic, three-dimensional model such as the ROMS application to Hood Canal. Typically, the model is run repeatedly with varying input parameters in an attempt to reduce error. Once this process has run its course (constrained to project goals, resources, and schedule), decisions are made to close the calibration process and run predictive scenarios with the model. In calibrating the biogeochemistry model of Kawase and Bahng (2012), the authors explored sensitivity to a large number of model parameters; however the exploration was by no means exhaustive or complete. Different combinations of parameters can give rise to nearly identical results (Kawase, pers. comm., 2011d).

If additional time and funding are available, supplemental data collection may improve model performance. Kawase and Bahng (2012) noted the absence of available particulate biomass data at the outside boundary and locations within the model domain. This data gap contributes to uncertainty in the model's total nitrogen budget, and it has hindered calibration of

parameters pertaining to the behavior of particulate matter in the model (Kawase, pers. comm., 2011d).

Once the model application matures to the point where model skill is sufficient to address specific management questions, the method of evaluating the questions can also introduce uncertainty. As noted above, the scenarios run for Hood Canal were highly idealized. Rather than defining tributary inflows to the model domain individually and assigning nitrogen levels using measured conditions, a uniform and constant concentration was applied to all tributaries. Another issue is the absence of explicit groundwater inflows and OSS loads to marine waters in both the calibration runs and scenarios.

More thorough documentation and review of the model is needed before recommending its use in a regulatory action, and improvements can also be made in quantitative accuracy and scenario construction. Nevertheless, the model is a very important analytical tool for dissolved oxygen problems, because it simultaneously and continuously accounts for processes that affect dissolved oxygen at a level of spatial resolution that captures the variable bathymetry of the Canal. All of the other approaches simplify the system to estimate marine loadings of nitrogen and dissolved oxygen impacts.

Uncertainty in the behavior of Lynch Cove also limits the application of the ROMS model to Lynch Cove for regulatory purposes. However, even given the uncertainty derived from model performance, the ROMS model corroborates screening-level aggregated model calculations that humans have a negligible impact on central Hood Canal dissolved oxygen or fish kills.

There is a general alignment of ROMS model estimates of DO impact and aggregated models (Steinberg) showing fractional contributions human versus marine.

Uncertainty in Nitrogen Loading Estimates

Based on the range of estimates in the studies reviewed and independent review comments, it is evident that there is substantial uncertainty in the reported estimates of both human and natural nitrogen loading. Each section of this review discusses uncertainty. The Monte Carlo analysis quantified the uncertainty in oxygen impact estimates due to uncertainties in the marine and human loading estimates.

It is not practical to list all of the assumptions and data gaps that contribute to the uncertainty in loading estimates here, but we offer the following examples of important areas of uncertainty:

- Uncertainty in estimation of marine nitrogen flux to the euphotic zone. Factors include limitations in methodology identified in this report and by the independent review panel and natural variability.

- Uncertainty in estimation of groundwater and OSS loading. Factors include variability in shoreline nitrogen samples, assumptions of the basin water budget,

and fate and transport of loadings at variable distance and depth relative to the receiving water.

- Uncertainty in the loading estimates from the statistical model used to analyze tributary loadings. Factors include selection of model parameters and model prediction error.

These uncertainties stem from a multitude of challenges in the overall analysis. Some sources of uncertainty can be quantified in a straightforward manner, such as the variability of sampling data. Some uncertainties, however, are more nuanced and difficult to quantify, such as the uncertainty associated with the omission of a potential nutrient sources or use of estimation methods that are over-simplified.

The available studies have not attempted to aggregate the uncertainty of individual component analyses (e.g., groundwater loading, watershed loading, marine fluxes, oxygen deficits) into an aggregate uncertainty range around the estimates for the dissolved oxygen impact. In this review, we have primarily focused on identifying all relevant information and reviewing the appropriateness of key analytical methods. These areas took priority over quantification of aggregate uncertainty. At the same time, the simplicity of the aggregated model approach allowed us to apply a simple Monte Carlo analysis to estimate the range of uncertainty in the estimates of human impact to dissolved oxygen in Lynch Cove.

Factors not Explored in Available Documents

Several other topics are relevant to the question of dissolved oxygen in Hood Canal but were not addressed in any published documentation.

Future Population Growth

All available studies of human-caused nitrogen loading are based on existing residential development and forest conditions. Subsequent studies should include analysis of future scenarios that account for expected changes in the watershed in the coming decades to inform management actions now.

Potential Effects of Sediment Enrichment

Lynch Cove sediments likely have a substantial influence on dissolved oxygen conditions and nutrient dynamics in this shallow area of Hood Canal. Over the long term (years to decades), sediments can be enriched if human releases of nutrients cause a significant increase in algae biomass, and this enrichment increases the sediment oxygen demand (SOD) on the overlying water. In addition, sediment enrichment affects the release of dissolved nutrients from sediments and denitrification in low oxygen waters near the bottom. No Lynch Cove or other sediment observations are provided in the studies reviewed

The ROMS model (Kawase and Bahng, 2012) included SOD as a model parameter but did not document the sensitivity of predictions to the selected input value for SOD. The model also included a function that assumed that particulate nitrogen reaching the bottom was remineralized as ammonium. Devol et al. (2011a) used a value for denitrification that cites Engstrom et al. (2009), which included sites within Hood Canal. However, Devol et al. (2011a) selected rates modified from Engstrom et al. (2009). Sheibley (pers. comm., 2011) describes denitrification at the groundwater-marine interface as a major loss mechanism, with rates of 1480 MT/yr (0 to 4800 MT/yr) for Lynch Cove. However, results of the flask studies have not yet been published.

Future studies should consider the potential for long-term enrichment of sediments, particularly in Lynch Cove, and the effects of enrichment on water quality.

Potential Effects of Hood Canal Bridge and Skokomish River Diversion

Located at the entrance to the Canal, the Hood Canal Bridge rests on large floating pontoons in the surface layer of the Canal. Natural circulation is characterized by the seaward flow of freshwater in the surface layer. No published studies evaluated the potential impact of the bridge on circulation (and ventilation). Management of the Skokomish River may also impact circulation (Newton et al. 2010). While the annual freshwater flow to the Canal from the river would be unchanged, the timing and location of flows from the Lake Cushman reservoir and hydroelectric facility could alter local, seasonal circulation. Similarly, Bremerton draws its water supply from the upper Union River watershed, which could affect streamflows. This potential influence was not described in any current Hood Canal materials.

The effects of the bridge and river management can be examined using mathematical models of circulation and water quality. It would difficult with existing data alone to tease apart these influences through data analysis alone because of the complex interplay of climate and human contributions. For example, the bridge was built in 1961 and the Skokomish River dams even earlier. Any reductions to circulation due to these factors also coincide with cyclic climate influences. Additional analysis would be warranted before ruling in or ruling out an influence of these factors on Hood Canal circulation.

Science Review Summary

This review incorporates information from published documents including draft and final reports, journal articles, and memoranda (see full citations in the References). Supplemental information was provided by Hood Canal researchers in meetings, conference calls, and email communications. Based on the best available information, we conclude the following:

1. Sediment cores establish that hypoxia occurred before European settlement, and oxygen levels were lower before 1900 than between 1900 and 2000. Shifts in organic matter associated with human watershed activities did not coincide with shifts in low oxygen conditions in Hood Canal. On the contrary, prior to the 1900s, low oxygen conditions

prevailed in a decadal pattern that is consistent with climate influences (Brandenberger et al., 2008 and 2011).

2. There is no compelling evidence that humans have caused decreasing trends in dissolved oxygen in Hood Canal. Climate effects cannot be ruled out, the decreasing trends are not unique to Hood Canal, and dissolved oxygen in Lynch Cove, where relative human influences are the largest, have increased and not decreased (Bassin et al., 2009).

    a. While the Hood Canal dissolved oxygen inventory shows a decline between the 1950s and 2000s, this pattern is also found in the Strait of Juan de Fuca and in other parts of Puget Sound and is not unique to Hood Canal. The decline is consistent with the decades-to-centuries climatic pattern that Brandenberger et al. (2008 and 2011) found in sediment cores.

    b. Lynch Cove, the area east of Sisters Point, is the part of Hood Canal where human contributions are most likely to influence dissolved oxygen concentrations. Lynch Cove dissolved oxygen concentrations show a statistically significant increase in concentrations between the 1950s and 2000s, counter to the decline in the Hood Canal dissolved oxygen inventory and counter to declines in other areas of Puget Sound.

3. Hood Canal phytoplankton growth is likely nitrogen-limited, so nitrogen mass loading to the euphotic zone directly affects dissolved oxygen conditions. The predominant overall source of nitrogen to Hood Canal is Pacific Ocean nitrogen entering as deep waters and entraining in the surface layer of Hood Canal. This dominant contribution of marine nitrogen holds throughout the year and throughout the main arm of Hood Canal and into Lynch Cove (Steinberg et al., 2010; Paulson et al., 2006; Devol et al., 2011a; and Kawase and Bahng, 2012).

4. Human activities have increased nitrogen loadings to portions of Hood Canal from tributaries and groundwater compared to natural conditions.

    a. Most of the tributary loading occurs between November and May, whereas severe hypoxia and fish kills have occurred in the summer months. Humans add to natural nitrogen loads directly through residential development, primarily OSS, and indirectly via red alders. Red alders fix nitrogen from the atmosphere and exist today in higher numbers than past years due to logging activities and slow regrowth of conifers.

        i. In Lynch Cove, tributaries contribute an estimated range of 40% to 80% of the total nitrogen loading from the watershed in the summer months. Red alders and residential development contribute approximately 50% and 33% of the tributary loading, respectively (Steinberg et al., 2010 and Richey et al., 2010).

    b. Nitrogen loadings entering Hood Canal from groundwater and shoreline on-site septic systems (OSS) are difficult to quantify because of variability in the natural and

human environment, limited sampling information, and uncertainties in fate and transport of subsurface wastewater.

i. Based on the best available estimates, loadings from shoreline OSS discharges in Lynch Cove range from 20% to 60% of the total nitrogen loadings from the watershed in the summer (see synthesis of Paulson et al., 2006, Georgeson et al., 2008, and Sheibley and Paulson, pers. comm., 2011).

5. Lynch Cove is the part of Hood Canal where human contributions have the greatest influence on marine dissolved oxygen. A synthesis of multiple studies indicates that human-caused discharges have an impact on dissolved oxygen levels in Lynch Cove that may approach the regulatory limit of 0.2 mg/L for waterbodies with naturally low dissolved oxygen. The available studies are inconclusive on compliance with water quality standards in this area due to substantial uncertainty in the methods, nitrogen sources, and averaging over time and space.

   a. For Lynch Cove, model simulations and aggregated models coalesce around an average summer impact ranging from 0.03 to 0.3 mg/L (see synthesis of Steinberg et al., 2010; Richey et al., 2010; Paulson et al., 2006; Georgeson et al., 2008; Sheibley and Paulson, pers. comm., 2011; Devol et al., 2011a; and Kawase and Bahng, 2012).

   b. Substantial uncertainty remains in the estimates of human impact to dissolved oxygen, particularly in estimates of marine nitrogen loadings to Lynch Cove. Other sources of uncertainty include the inability of simple models to represent complexity of the system, insufficient spatial and temporal data, and factors omitted from analysis. In addition, the maximum impact cannot be addressed quantitatively in Lynch Cove due to limitations of currently available models. Due to the uncertainty and analysis limitations, it is not possible to definitively assess whether the water quality standards are currently violated by human-caused nitrogen loads with the available information.

   c. Variability in groundwater nitrogen concentrations is a key area of uncertainty and concern. Groundwater quality is generally good, but a small number of seeps have extremely high nitrogen concentrations. These seeps have a disproportional influence on the overall human loading of nitrogen. Pollution Identification Program (PIC) efforts have shown the ability to drastically reduce these concentrations through improved OSS management.

6. Researchers have unraveled the conditions that lead to fish kills. Fish kills occur due to a cascade of natural events. Dense marine water enters Hood Canal and lifts water with low oxygen levels toward the water surface. As river inflows decline during the dry season, the freshwater cap on the surface thins. Southwest wind events push this thin cap to the north, which allows low-oxygen water beneath it to surface rapidly. In a matter of hours, oxygen levels rapidly decrease in the sensitive nearshore regions of southwestern Hood Canal

(Newton et al., 2011c, and Kawase and Bahng, 2012), which causes the fish kills at Sund Rock near Hoodsport.

a. Because of the large-scale nitrogen loadings from marine waters, human-caused nitrogen discharges have almost no effect on dissolved oxygen in central Hood Canal. In the main arm of Hood Canal (from Lilliwaup to Potlatch), where fish kills have been reported, initial model simulations indicate very small impacts (maximum impact less than 0.04 mg/L) to dissolved oxygen from human-caused nitrogen loadings (Kawase and Bahng, 2012).

b. In addition to the episodic events at Sund Rock near Hoodsport, chronically low dissolved oxygen in Lynch Cove (east of Sisters Point) in the summer can stress or kill marine organisms, particularly bottom-dwelling organisms (Newton et al., 2011c).

c. Best available information indicates that the subsurface seaward outflow has an insignificant effect on fish kills at Hoodsport.

Recommendations for Future Technical Work

Given the substantial limitations and uncertainties in the work to date, additional study of human impacts to Lynch Cove may assist planning and regulatory activities in the watershed in the future. We recommend the following actions to improve the estimates of human impact on dissolved oxygen:

1. Develop Refined 3D Biogeochemical Model of Lynch Cove

   Develop a model that more fully represents the physical characteristics of Lynch Cove (e.g., shallow areas, wetting/drying) and reasonably predicts the observed vertical structure in dissolved oxygen, nitrogen, and chlorophyll a. This may require additional observations to describe key processes such as sediment-water interactions.

2. Organize, Archive, and Share HCDOP Data

   The extensive raw data from tributary and marine sampling collected by HCDOP is not readily available to researchers outside HCDOP. This data should be organized and archived on CDs and or a publicly-available website so that other organizations and individuals can conduct additional modeling and analysis.

3. Re-analyze Marine Nitrogen Flux

   While development of a refined biogeochemical model is the top priority, re-analysis of the nitrogen flux calculations for the aggregated 2-layer model of Lynch Cove would

provide valuable screening-level information for comparison to estimates of a new 3D model recommended above. This task was recommended by the independent review panel based on concerns about the methodology used to date.

4. Continue Shoreline Seep/Groundwater Monitoring

   Shoreline sampling of seeps provides important information about the quality of groundwater entering Lynch Cove from the shoreline. Continued investigation of the sources of high nitrogen seeps will improve estimates of shoreline OSS loadings and identify failing or poorly managed OSS systems.

5. Analyze De-Nitrification at Sediment Interface

   Several researchers have noted that low oxygen conditions at the bottom of Lynch Cove are conducive to de-nitrification of groundwater seeps. This process may be significantly reducing the nitrogen flux to Lynch Cove from shoreline OSS. A better understanding of the role of de-nitrification in the Lynch Cove nutrient budget would help estimate the overall budget of nitrogen sources and sinks in the watershed.

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Glossary

Advection	The transport of water due to the bulk movement of water parcels, such as through tidal exchanges.
Aggregated model	Simplified representation of complex system that is used for screening-level assessments
Ammonium	Form of nitrogen present in natural waters and effluent from wastewater treatment systems. Part of the dissolved inorganic nitrogen. Ammonium is the form of ammonia present at pH values typical of natural waters and wastewater effluent.
Cfs	Cubic feet per second
Clean Water Act	Federal act passed in 1972 that contains provisions to restore and maintain the quality of the nation's waters. Section 303(d) of the Act establishes the Total Maximum Daily Load (TMDL) program.
DIN	Dissolved inorganic nitrogen, which is the sum of nitrate, nitrite, and ammonium. The forms of nitrogen most readily available for biological processes.
DON	Dissolved organic nitrogen
ECY	Washington State Department of Ecology
Eddy diffusion	The mixing of substances within waters due to turbulent diffusion processes
EPA	U.S. Environmental Protection Agency
Estuarine circulation	Water circulation pattern that results from a net inflow of marine water near the bottom of the water column coupled with a net outflow of fresher water near the surface.
Euphotic zone	Surface water layer where light fuels photosynthesis
HCCC	Hood Canal Coordinating Council
HCDOP	Hood Canal Dissolved Oxygen Program
Hood Canal	Marine waters south of Foulweather Bluff and Tala Point. Sometimes defined by the sill or Hood Canal floating bridge
Isopycnal	Line or surface of constant density, which in marine waters includes the effects of both salinity and temperature
Isotach	Line or surface of constant velocity
Loading	The input of pollutants into a waterbody
Lower Hood Canal	Portion of Hood Canal generally defined as the region east of Sisters Point
Lynch Cove	Portion of Hood Canal defined as either the region east of Sisters Point (equivalent to Lower Hood Canal) or the shallowest portions of Lower Hood Canal
m^3/s	Cubic meters per second
Marine water	Salt water
Mg/L	Milligrams per liter, equivalent to parts per million
Mo	Month
MT	Metric tons, equivalent to 1,000 kg
Nitrate	Form of nitrogen present in natural waters and effluent from wastewater treatment systems. Part of the dissolved inorganic nitrogen.

Nitrite	Transitional form of nitrogen present in raw wastewater but not typically found in natural waters or effluent from wastewater treatment systems. Part of the dissolved inorganic nitrogen.
Nonpoint sources	Pollution that enters any waters of the state from any dispersed land-based or water-based activities, including but not limited to atmospheric deposition, surface-water runoff from agricultural lands, urban areas, or forest lands, subsurface or underground sources, or discharges from boats or marine vessels not otherwise regulated under the NPDES program. Generally, any unconfined and diffuse source of contamination. Legally, any source of water pollution that does not meet the legal definition of "point source" in section 502(14) of the Clean Water Act.
NPDES	National Pollution Discharge Elimination System. National program for issuing, modifying, revoking, and reissuing, terminating, monitoring, and enforcing permits, and imposing and enforcing pretreatment requirements under the Clean Water Act. The NPDES program regulates discharges from wastewater treatment plants, large factories, and other facilities that use, process, and discharge water back into lakes, streams, rivers, bays, and oceans.
Nutrient	Substance such as carbon, nitrogen, and phosphorus used by organisms to live and grow. Too many nutrients in the water can promote algal blooms and rob the water of oxygen vital to aquatic organisms.
OSS	Onsite sewage systems, including but not limited to septic systems
PDO	Pacific Decadal Oscillation
PIC	Pollution Identification and Correction
PNNL	Pacific Northwest National Laboratory
Point source	Sources of pollution that discharge at a specific location from pipes, outfalls, and conveyance channels to a surface water. Examples of point source discharges include municipal wastewater treatment plants, municipal stormwater systems, industrial waste treatment facilities, and construction sites that clear more than 5 acres of land.
Ppt	Part per thousand, equivalent to mg/L
Puget Sound	The marine waters south of Admiralty Inlet
Pycnocline	The vertical marine layer where the density changes the most quickly. Generally the pycnocline is shallower than the euphotic zone.
ROMS	Rutgers Ocean Model
SOD	Sediment oxygen demand
TDN	Total dissolved nitrogen
TN	Total nitrogen
Total Kjeldahl nitrogen	Form of nitrogen that includes ammonium and dissolved organic nitrogen if the sample is filtered, but also particulate organic nitrogen if the sample is not filtered
Ug/L	Micrograms per liter, equivalent to parts per billion
USGS	U.S. Geological Survey
UW	University of Washington
Watershed	A drainage area or basin in which all land and water areas drain or flow toward a central collector such as a stream, river, or lake at a lower

	elevation.
Yr	Year
Zero isotach	The water column layer that defines the depth of no motion, where the surface layer has a net seaward flow and the lower layer has a net landward flow